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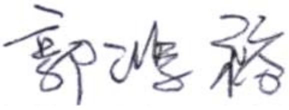
序言

非點源污染爲水污染的來源之一，其導致河川、湖泊、地下水及水庫等表面水體及地下水體的水質不佳。各種非點源污染中，農業活動爲其造成此種污染的因素之一，尤其是不當的施肥行爲使環境中營養鹽的累積，營養鹽中以氮和磷造成水生植物、細菌和藻類大量繁殖，導致魚貝類無法生存的優養化現象的污染情況。目前，已發現台灣地區部份的非點源污染來自於農業活動，有鑑於此，行政院農業委員會與中華土壤肥料學會共同舉辦「農業非點源污染」研討會，共邀請九位國內外學者提供有關國內外農業非點源污染的情形，貢獻其研究經驗，此研討會主要目的在於了解台灣來自農業施肥之非點源污染概況，並藉由歐盟經驗交流，討論國內未來可能的監測計畫或防治對策。



謹識

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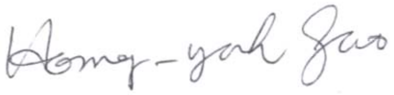
Preface

Non-point pollution is one of the source of water pollution, leading quality degradation of surface water like rivers, lakes and reservoirs and ground water. Agricultural activities partly account for non-point source pollution, especially, improper fertilizer use, which causes nutrient accumulation in the environment. The two nutrients most affecting water qualities are nitrogen and phosphorus, which are increasing in aquatic plants, bacteria and algae blooms. Consequently, depletion of dissolved oxygen takes place and results in death of fishes and shellfish. This is termed as *eutrophication.* At present, nonpoint pollution resulted from agriculture are found some area in Taiwan. Hence, *Agricultural Research Institute* and *The Chinese Society of Soil and Fertilizer Sciences* co-organize the “Nonpoint Pollution from Agriculture Conference’’. With nine professors both domestic and from abroad invited and participating in the conference, their research achievements and experience on nonpoint pollution from agriculture will be shared and contributed. The main purpose of this conference is to better understand present status of non-point pollution from agriculture in Taiwan and to exchange experiences with European Union as well as further discuss plans for monitoring and preventing non-point pollution from agriculture activities in future.



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Diffuse Pollution by Livestock Production Systems and Manure Management

Oene Oenema[1,3] and Harald Menzi[2]

Abstract

Manure management systems are conducive to nutrient and carbon losses, but the magnitude of the loss highly depends on the nutrient element, the manure management system and the environmental conditions. The aim of this paper is to contribute to the understanding of the diffusive nutrient losses. Firstly, we discuss the main mechanisms loss path ways of nutrients from manure management systems to groundwater, surface waters and air, using some data from the European Union (EU-27). Secondly, we discuss possible measure to decrease diffuse pollution and to increase the environmental performance of livestock production and manure management. Finally, we give some policy recommendations.

In general, losses from animal manure decrease in the order: C, N >> S > K, Na, Cl, B > P, Ca, Mg, metals. The total N excretion by livestock in EU-27 was ~10400 kton in 2000. About 65% of the total N excretion was collected in barns and stored for some time prior to application to agricultural land. Almost 30% of the N excreted in barns was lost during storage; approximately 19% via NH_3 emissions, 7% via emissions of NO, N_2O and N_2 to the air and 4% via leaching and run-off from manure storage systems. Another 19% of the N excreted in animal housing systems was lost via NH_3 emissions following the application of the manure to land. The results indicate that maximally 52% of the N excreted in barns was effectively recycled as plant nutrient. Various emission abatement measures can be implemented and have been implemented already in some Member States to reduce the emissions of NH_3 and N_2O, and the leaching of N and P. There is large scope to decrease diffuse pollution by livestock production systems through policy and measures and improved management, but the economic costs are relatively high.

Keywords: ammonia, diffusive losses, environmental policy, European Union, manure management, nutrients, nitrogen, nitrate, nitrous oxide, phosphorus

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1. Introduction

Livestock production systems exert various influences on the environment. The influences greatly depend on the livestock production system itself, the management and the environmental conditions. Much of the influence of livestock systems on the environment occurs via its effects (direct and indirect) on land use (changes) and nutrient element cycling. These effects have increased greatly over the last decades, particularly in response to the current trends in livestock production: up-scaling, intensification, specialization and regional conglomeration (Foley *et al*., 2005; Naylor *et al*., 2005; Steinfeld *et al*., 2006).

Though the global picture of the impact of the intensification of livestock production on the environment has become clearer during the last years, many of the complex relationships between livestock production systems and environmental impact are not yet well-understood. This holds especially for the fate of nutrients and carbon from the livestock manure, as little is known about the management of manure in practice. There is a huge diversity in livestock production systems, and there is a wide variety in the management of these systems. This diversity in systems and in management contrasts with the small number (14) of types of domestic animals in the world, and especially with the small number of types of the most important categories (from an economic point of view): cattle, pigs and poultry (Diamond, 2002). Though livestock consumes less than 3% of the global net primary production (Smil, 2002), its contribution to the global burden of ammonia (NH_3), methane (CH_4) and nitrous oxide (N_2O) in the atmosphere ranges between 10 and 40% (Bouwman *et al*., 1997; Oenema and Tamminga, 2005). Globally, livestock excretes about 100 Tg (range 70-140 Tg) nitrogen (N) per year, but only 20-40% of this amount is recovered and applied to crops (Sheldrick *et al*., 2003; Oenema and Tamminga, 2005). The remainder is dissipated into the environment. The estimated amounts of phosphorus (P) and potassium (K) in livestock manure are 1.5 and 3 times the current amounts of P and K in mineral fertilizers (Sheldrick *et al*., 2003), but only a fraction of manure P and K is efficiently utilized. In general, the potential for recycling the nutrients from animal manure effectively is not fully realized and seems difficult to realize fully without drastic improvements in manure management, use of environmental technology, and restructuring conglomerations of landless livestock production systems (Sims *et al*., 2005). The total amounts of N and or P excreted per unit surface area are proper indicators for the environmental pressure of livestock production systems. Fig. 1 shows the mean spatial distribution of P from livestock in the world. Clearly, livestock is unevenly distributed over the globe. There are large concentrations in parts of China (especially pigs and poultry), Europe (cattle, pigs and poultry), India (cattle), and to a smaller extent America (cattle, poultry pigs).

The aim of this paper is to contribute to the understanding of diffusive nutrient losses from livestock production and manure management to the environment. Firstly, we discuss the main mechanisms loss path ways of nutrients from manure management systems to groundwater, surface waters and air. Secondly, we discuss possible measure to decrease diffuse pollution and to increase the environmental performance of livestock production and manure management. Finally, we give some policy recommendations.

2. Nutrient losses from manure and its environmental impacts

Elements are lost from livestock manure management systems at different rates. Roughly, the descending order is as follows: C> N >> S > K, Na, Cl, B > P, Ca, Mg, Fe, Mn, Cu, Zn, Mo, Co, Se, Ni. This order is related to the reactivity, speciation, solubility and fugacity of the nutrient element species and their mass fractions in manure. The double mobility of C, N and S, in soluble waterborne compounds as well as in gases, makes their cycling and loss pathways much faster and more complex than those of the mineral elements. Carbon is released from manure in gaseous forms (mainly CO_2 and CH_4), in dissolved forms as inorganic and organic C (ΣHCO_3, DOC), and as particulate matter (via run-off). Nitrogen is also released in gaseous forms (mainly NH_3, N_2, N_2O, NO), in dissolved forms as inorganic and organic N (NO_3^-, NH_4^+, DON), and as particulate matter (via run-off). Sulfur is mainly lost via volatilization of sulfides (H_2S) and sulfur oxides (SO_2) and the leaching of sulfate (SO_4^{2-}) and particulate matter.

The mineral elements can be lost via leaching in dissolved form or via run-off in particulate matter at various stages of the feed – animal – manure – soil – crop chain (Fig. 2). Leaching of effluents from ensiled feed and manure from animal houses and storage systems is negligible when the ground is sealed and the drainage water collected. High concentrations of, for example, K, Cl and NO_3^- in wells near farm houses are often seen as evidence of leaching losses from manure heaps on top of unsealed soil. Potassium (K), sodium (Na), chloride (Cl) and boron (B) have high solubility in water, and the main loss pathway is via leaching. Finally, the elements P, Ca, Mg, Fe, Mn, Cu, Zn, Mo, Co, Se and Ni have low mobility because of their low solubility and high reactivity with soil constituents. Some of these elements (Fe, Mn, Se) are redox-active and more mobile in chemically reduced forms. Some elements may form complexes with dissolved organic carbon and inorganic anions and thereby increase there mobility.

Few studies have quantified nutrient losses from animal manure storage systems via leaching and run-off. The available information suggests that N leaching losses from slurries in unsealed

lagoons in EU-27 are in the range of 5% of the N in slurry when covered, and in the range of 10% when uncovered. Similarly, N leaching losses from solid manure in unsealed heaps are in the range of 2% of N in the manure when covered, and 5 to 10% when uncovered. Needless to say, the uncertainty in these estimates is high.

Current environmental concerns relate mainly to gaseous emissions of NH_3, N_2O and CH_4 from manure management systems to the atmosphere, and the leaching of NO_3^- to groundwater and N (NO_3^-, NH_4^+) and P (ΣPO_4, particulate matter) from manure management to surface waters. The excessive accumulation of Cu and Zn in soils following heavy applications of pig and poultry manures has received considerable attention in Europe in the 1980s and 1990s, but is now less a concern following governmental regulations on the additions of Cu and Zn in animal feed, and on manure application rates. However, this is not yet the case in many countries in Asia and Latin America.

2.1 Pollution of fresh water ecosystems

The effects of livestock wastes on aquatic ecosystems are basically the same as in case of household and municipal wastes. These effects can be summarized by eutrophication and toxification, which lead subsequently to ecosystem changes. The effects follow from the presence of degradable organic matter, nutrients (mainly N and P), heavy metals (mainly Cu and Zn), and possibly pathogenic micro-organisms, hormones and antibiotics in the wastes.

Eutrophication has been among the main water quality issues since decades. It is caused by excess loads of nutrients (mainly N and P) and organic matter to surface waters. It leads to excessive growth of phytoplankton and subsequently to decreases of water clarity and biodiversity, oxygen depletion, fish kills, and odor problems, while harmful toxic algal blooms may develop more frequently (Rabalais, 2004). Agriculture contributes roughly 50% to the total loading of surface waters with N and P, but the specific contribution of animal agriculture is unknown. Eutrophication can be exacerbated by the natural growth of toxic cyanobacteria (sometimes incorrectly referred to as blue-green algae). These micro-organisms are commonly occurring in all water bodies, but are particularly abundant in tropical and subtropical climate, and are notorious for their production of toxins that are detrimental or even deadly for human and animals incidentally consuming water with high concentration of cyanobacteria.

Pathogens associated with livestock and belonging to such microbial groups as viruses, bacteria, protozoa and helminths are known to penetrate into aquifers and surface water bodies. They present a real threat to humans and animals alike These threats are taken more and more

seriously particularly in the light of recent SARS and avian influenza epidemics. These epidemics though viral by nature, demonstrate and accentuate an intrinsic potential danger of all groups of microorganisms emanating from livestock through wastes.

2.2 Pollution of marine coastal ecosystems

Through the transport of nutrients via rivers, and the deposition of ammonia-N originating from manure, livestock production also contributes to euthrophication of marine coastal ecosystems. The flux of nutrients from land to sea has increased since the 1960s by over 3-4 time, and nutrients originating from livestock production played a prominent role in this. Eutrophication stimulates the growth of marine algae's, be it microalgae (which lead to 'red and brown tides') or macroalgae (seaweeds). Transfer of pathogens from livestock wastes to sea is not an uncommon occurrence and results in temporary closures of marine aquaculture farms due to health risks resulting from elevated coliform counts. Furthermore, eutrophication-related impacts on sensitive estuaries, coast zones and coral ecosystems, which are among the most productive ecosystems on the planet, lead to loss of biodiversity which may lead to coastal erosion. This may coincide with decreased commercial harvests of marine products and necessitates rehabilitation associated with high costs. Eutrophication of marine ecosystems is a global concern.

2.3 Groundwater pollution

The leaching of nitrate to groundwater is considered a problem because when used as drinking water, the nitrate is consumed by humans and may cause methemoglobinemia (blue baby disease). Livestock excrements contribute to nitrate leaching, indirectly through crops and grassland highly fertilized with manure and when grazing livestock is fed protein-rich diets. The nitrate is generated by the nitrification of ammonium from fertilizer and excreta. The nitrate may also originate from leakages of manure storage systems (Oenema *et al*., 2007). Nitrate pollution of groundwater impacts water resources available to communities leading to health and social problems. However, there is still debate about the actual toxicity of nitrates in drinking water consumed by humans.

2.4 Nutrient accumulation in soil ecosystems

Plants need N, P, K and 10 other nutrients for their growth and functioning. In crop production, nutrient uptake per crop from the soil may range from 50 to 300 kg ha^{-1} for N, from 10 to 40 kg ha^{-1} for P and from 50 to 400 kg K ha^{-1} for K. In double and triple cropping systems, uptakes of N, P and K may be even higher. These wide ranges reflect differences between crop types, and soil fertility and fertilization levels (Laegreid *et al*., 1999). Theoretically, the whole nutrient requirement of the crops could be met by livestock manure. However, in practice this is usually not feasible, because

the ratio of the different nutrients in manure is not the same as that for crop requirements and because it sometimes is not possible to spread manure when there is a particular N demand of the crops (e.g. during a wet spring in Europe or during the rainy season in the tropics). When application rates exceed the uptake capacity of the crop, surplus nutrients accumulate in the soil or are lost to the wider environment, depending on the mobility and reactivity of the nutrients and the environmental conditions (rainfall, topography). Phosphorus and also zinc and copper are strongly sorbed to the soil, and therefore have low mobility and hence tend to accumulate in soils. In contrast, nitrogen and sulfur have high mobility and tend to dissipate into the wider environment. Calcium, magnesium and potassium take an intermediate position.

For P, the strong accumulation in the top soil will be a concern for aquatic ecosystems when topsoil with a high concentration of P will enter surface waters via erosion. High K concentrations in soils may lead to a high K uptake by the crop, which may lead to grass tetany (or hypomagneseamia) in grazing dairy cows. Furthermore, high amounts of K can cause an increase of soil salinity with negative impact on crop yield and soil fertility. Lowering high P and K levels in soils may take many years; it requires that the application rates of animal manure (and fertilizers) are decreased strongly.

2.5 Soil pollution with heavy metals and organic compounds

Some heavy metals are used in livestock feed in amounts surpassing the essential animal requirements. Especially copper (Cu) and zinc (Zn) are often used in relatively high doses in pig production because of their growth promoting and therapeutic effects. Other heavy metals that are sometimes added to animal feed are chromium (Cr), cobalt (Co) and, arsenic (As). Well over 90 % of the heavy metal inputs to feedstuffs are excreted by the animal. Manure, especially pig and to a lesser extent poultry manure, therefore contains relatively high concentrations of heavy metals. The Cu and Zn content commonly found in pig slurry in Europe already leads to an accumulation of these metals in the soil. Excessive manure application to soils will further increase this accumulation. As heavy metals are not degraded or lost, it is only a question of time until a level is reached where the soil fertility is harmed and crop yields are affected. Especially in areas with fast growing livestock production, high livestock density and weak environmental legislation, heavy metals are increasingly gaining importance as a threat to the environment.

2.6 Emissions to the atmosphere

Apart from the effects of N from livestock manure in groundwater and surface waters, livestock manure also contributes to volatilization of ammonia into the atmosphere and to increased

atmospheric N deposition. Atmospheric N deposition is especially a problem in vulnerable ecosystems sensitive to high N input (Galloway *et al*., 2003). Typically, 20 to over 50 % of the N excreted by housed livestock is lost through ammonia volatilization (Oenema *et al*., 2007). Thus, livestock production typically contributes 70 to over 90 % of the total ammonia emissions (Bouman *et al*., 1997). Furthermore livestock production (especially ruminants) and manure management are an important source of methane (CH_4) emissions and also contribute to nitrous oxide emission, which are both important greenhouse gases.

Fig. 3 shows the spatial distribution of N losses from agriculture via emissions of ammonia and nitrous oxide to the atmosphere and the leaching of nitrate to groundwater and surface waters. It shows that the patterns of emissions are rather similar. Hot spots of emissions are related to regions with intensive agriculture, mainly in The Netherlands, Belgium, Denmark and parts of France, Italy and United Kingdom.

3. Improving the performance of livestock farming

There are various options to improve the environmental performances of intensive livestock production systems. These options can be divided in measures related to improving (i) the management, (ii) the technology and (iii) the system or structure of the production – consumption chain. Management-related options are based on improved insights and knowledge (software) and are usually relatively cheap. Technological measures (hardware) are often part of management and system related measures; they require capital investments. Structural measures deal with fundamental adjustments of the structure of the whole ‘supply – production – processing – marketing – retail – consumption chain (orgware)’, and therefore not easy to implement by single farmers. From a cost-effectiveness point of view, management-related measures must be considered first. While doing so, technological measures have to be considered. Structural measures must be considered when management and technological measures do not bring solutions.

Improving the performance requires farm specific analyses of the feed – animal – animal produce – animal manure – crop production chains. The weakest part of this chain in terms of agronomic, economic and environmental performances usually determines the best options for improvement, in terms of effectiveness and efficiency. A series of consecutive steps have to be considered (Oenema and Pieterzak, 2001; Oenema and Tamminga, 2005). The weakest part of the chain in intensive livestock production systems commonly is the storage and utilization of the animal manure produced in these systems. Therefore, focus in this chapter is on improving ‘animal

manure management'. However, we start our discussion with the producers of manure; the livestock component.

3.1 Improving livestock management

The amounts and composition of the animal excrements are directly dependant on the number, type (species), productivity and feeding of the animals, while the total amounts of manure to be handled strongly depends on the amounts of water and litter used in animal housing. Therefore, the following aspects of livestock management are relevant for manure management:

- The higher the productivity and the more efficient the feed conversion, the lower the amount of manure produced per animal and per unit product. This is a key option for improving the economic and environmental performances of livestock farms. It requires improvement of the genetic potential of the herd, phase feeding (precision feeding) and proper housing of the animals. However, improved productivity might not diminish the load to the environment if it is achieved with commercial feed that substitutes local resources.
- The protein and P concentrations in the diet should be limited to the minimum requirement of the animals, so as to minimize the amount of N and P in the manure and to maximize resource use efficiency.
- Concentrations of heavy metals in animal feed must be limited to the minimum requirement of the animals, to limit heavy metal concentrations in the manure and to prevent accumulation in the soil.

3.2 Improving manure management

Manure contains valuable nutrient elements and these must be recycled via application to cropped land. Proper manure recycling and application to crops means that all the livestock excreta must be collected, stored with minimal losses until the nutrients they contain are needed by the crop, transported to the field, applied evenly and in doses corresponding to the nutrient requirements of the crops. This implies minimizing the amount of water being collected together with the slurry, sufficient leak-tight storage capacity, and minimizing NH_3 volatilization through covered slurry stores and low emission spreading techniques.

There is a wealth of information about improving the recycling of manure nutrients effectively in crop production through proper implementation of management and technological measures. Some of this information is derived from experiments, but some also from practice. For example, the default N fertilizer and P fertilizer values of pig slurry in Denmark are 70 % and 100 %, indicating that 70 % of the N and 100 % of the P in the pig slurry must be accounted for in the

nutrient balance, or in other words be considered as effective as high quality N and P fertilizers. Moreover, a fraction of the residual N may be utilized in subsequent years following the mineralization of the organically bound N in the manure. However, in many countries the mean utilization of manure nutrients is much lower than in Denmark, in part because of poor management and technology, in part also because of system constraints, as further discussed below.

3.3. System improvements and structural adjustments

A fundamental flaw of intensively (industrialized) livestock production systems is the large-scale geographic uncoupling of feed production from animal production. Landless intensive livestock production systems tend to agglomerate near large markets, and import essentially all animal feed from elsewhere, often from abroad. By doing so, nutrients in soils of crop producing area are depleted while livestock producing areas become enriched with nutrients. Nutrient depletion in soils can be remediated via fertilizer applications, but nutrient enrichment can not be solved without carrying the nutrients away from the site of manure production. Basically, nutrient depletion and nutrient enrichment are naturally occurring processes, but the current scale and intensity of nutrient depletion and nutrient enrichment associated with intensive livestock production in some regions is unprecedented, and is likely to increase further.

3.4 Towards sustainable livestock production systems

The question emerges 'are current agglomerations of intensive livestock production systems sustainable in the long-term'? If the answer is no, the next question is 'what solutions are available'?

The answer to the first question is largely 'no', although the answer depends in part on the scale and intensity of the agglomerations and on the perception of 'sustainability'. The sustainability is less of an issue when the manure from intensive livestock production systems can be disposed off properly on nearby crop land, compared to systems that would have to transport the manure hundreds of kilometers away, and therefore dump or discharge the manure locally. Further, the term 'sustainability' is value-laden; basically it deals with the quality of life and about the possibilities to maintain that quality in future. It depends on the notions in the society about the quality of life and its geographic distribution, as well as on the scientific understanding of the functioning of humans and natural ecosystems on earth. Current scientific understanding is that the disruption of the natural nutrient cycles through agglomerations of industrialized livestock production systems is not sustainable. It leads to

massive eutrophication and loss of biodiversity in livestock producing areas and to soil depletion and degradation in animal feed producing areas. Note also that the easily accessible phosphorus rock reserves will be depleted in 50 to 100 years (Laegraid *et al.*, 1999; Smil, 2000). Further, society may not accept the local dumping and discharge of manure nutrients from industrialized livestock farms and may call for governmental policy and measures as is the case in the European Union (Romstad *et al.*, 1997). Restrictions on dumping and discharge of manure will increase the cost of production and will force farmers of intensive livestock production systems to search for innovative solutions.

What sort of innovative solutions are available? When management and technological measures are insufficient to solve the nutrient unbalance, fundamental changes in the structure of the production – consumptions chain are needed. Such structural adjustments may lead to transitions of the agricultural systems. In this case, the structural adjustments have to correct the fundamental flaw associated with industrialized livestock production systems. Roughly, four lines of thoughts of structural adjustments are possible:

1. High-tech manure processing, yielding marketable manure products that can be stored, transported and applied to crop land elsewhere within economic, environmental and social constraints. However, manure processing techniques that yield marketable manure fractions (for example, separation with anaerobic digestion, composting and/or drying of the solid fraction, and ultra-filtration and reverse-osmosis of the liquid fraction) are expensive. As a result, intensive livestock production systems with manure processing technology may have to be combined and integrated with intensive crop production systems, food processing and heat and element re-cycling units in high-tech agro-production parks. Because of their high-tech, knowledge-intensive and cost-effective and environmental friendly re-use and recycling of wastes, these systems may have a comparative advantage relative to conventional livestock farms that dispose off their manure inappropriately. A few of these agro-production parks are now under development in China, India and The Netherlands; its success in practice still needs to be proven.
2. Spatial zoning and planning of intensive livestock production systems to such an extent that the livestock density within a region allows the proper disposal of all untreated animal manure on nearby crop lands within environmental, economic and social constraints. Evidently, this spreading of intensive livestock production systems over larger areas goes against the trend of agglomeration to benefit from site-specific conditions (markets, logistics, marketing, extension

and research services, etc.). Such spatial zoning and planning has been introduced via (tradedable) pig and poultry quota per farm combined with ceilings of the maximum number of pigs and poultry per region in The Netherlands, so as to limit the further expansion of the intensive livestock pig and poultry sector. However, such an animal quota system may also introduce (economic) side-effects. To be maximally effective, a quota system should be implemented when livestock stocking density in a region does not yet exceed critical limits.

3. A greater structural change occurs when animal production will be made land-bound again. It will connect crop production to animal production locally and regionally. The large-distance transport of animal feed will be replaced by large-distance transport of meat, while the small-distance exchange of animal feed and animal manure allows effective recycling of nutrient elements at low cost. Evidently, achieving such a structural change requires a major transition, as the whole production – consumption chain has to be relocated. It may take generations and it requires strategic international agreements, as it relates to food security, entrepreneurship, stewardship and trust.
4. Even greater structural changes will be brought about through a transition in protein production and consumption, shifting from meat to plant proteins, or from ‘pigs to peas’ (Fig. 4). A shift from a diet with relatively large proportions of meat protein to plant-derived protein products appears to be environmentally more sustainable, technological feasible and socially desirable (Aiking *et al*., 2006). Such a transition will also contribute to releasing pressure on freshwater resources and will decrease the energy-intensity of food production. Again, such a protein transition will have to be implemented at the global level. A full transition will more than halve the needed agricultural land and will more than halve the euthrophication associated with food production.

Clearly, these 4 lines of thought encompass transitions of the whole food production (and consumption) chain. Fundamental to transitions is the change in structure and characteristics of (part of the) society, its large scale, and its large impact.

3.5 Governmental policy measures

Governmental policy and measures aim at changing humans’ behaviour; they are meant to correct unwanted behaviour and/or to stimulate desired behaviour. Environmental policy and measures in agriculture aim at changing farmers’ behaviour and thereby aim at improving the environmental performance of the farms. The changes brought about by the change in farmers’ behaviour can be categorized in three types of effects, namely (i) managerial changes, (ii)

technological changes, and (iii) structural changes. A distinction between these types of changes is important as it will contribute to increased understanding of the effects of governmental policy and measures. Managerial changes involve investments in the ‘software’ of the farm, in the execution of day-to-day farm practices. These changes require a better integration of farmers’ knowledge and scientific knowledge through interactive training, education and demonstration activities. And they require suitable platforms for interaction between farmers, researchers and industry. Technological changes involve investments in the ‘hardware’ of the farm, in the buildings, machines, and equipment. These investments are often costly, and usually are only made when combined with up-scaling (enlargement of the farm) to make the investment cost-effective, or following suitable incentives (subsidies, tax reduction, fines). Structural changes concern changes in the orgware of agriculture, in the type (land-based versus landless systems, specialized versus mixed systems), size and location of farming systems, in the vertical (from suppliers to consumers) and horizontal (co-operation among farmers) organisation. Structural changes involve changes in the relative importance of production factors and resources (land, labor, capital, energy and management), and may involve changes in ownership of farms and farm land, and in the organization of farmers and the institutionalization of farmers’ organizations.

It is important to note that the way in which policy is designed and implemented is subject to changes over time. Many developed countries may rely on the functioning of classical-modernist state structures that have strong legitimating from the people, although this automatic trust and legitimating has been decreasing during the last three decades. Notably, the polity and the role of knowledge in the policy-making process are changing. Polity can be understood as the political setting in which policy-making takes place. During the last decades, stakeholders like environmental NGO’s and consumer organisations increasingly have a say in the policy making process. Further, the role of scientific knowledge is changing. Scientific knowledge is crucial for developing sound environmental policy and measures, but at the same time it is clear that science can no longer pretend to hold all facts and provide clear blueprints for policy. In environmental issues the processes involved are so complex and heterogeneous that uncertainty in policy outcomes is high.

There are three policy approaches to design environmental policies that vary in the extent of governmental coercion, namely (i) hierarchical control, (ii) interactive steering and (iii) self-governance. In the policy approach of hierarchical control it is the (central) government that initiates, executes and controls the policy. The underlying reasoning is based on expert knowledge.

The policy instruments are laws, fines and subsidies. The interactive steering approach leaves a greater role for private parties. The approach is based on the premise that necessary societal changes can only be successfully started if institutionalized private parties are involved in the policy-making process. Agreements are written down in covenants and contracts. Interactive steering can be successful in situations of great complexity and important dependency relationships. Self-governance takes the private company or organisation as a point of departure. Self-governance can therefore be defined as "private regulation within a public framework". The government steps aside and leaves execution and sanctioning to a system of control on self-control. The government does state the structure within which private parties can operate. Self-governance works best when the own interest of the individual company overlaps with governmental interests.

Once the rationale, approach and objective of the policy have been formulated, instruments have to be selected to implement the policy in practice. Policy instruments can be divided into three main categories: (i) regulatory or command-and-control instruments, (ii) economic or market-based instruments and (iii) communicative or persuasive instruments. These are briefly described below.

Regulatory instruments (regulation) involve a restriction on the choice of agents, methods and actions. Regulations are compulsory measures imposing requirements on producers to achieve specific levels of environmental quality, including environmental restrictions, bans, permit requirements, maximum rights or minimum obligations. They are the most common policy measure used in EU-environmental policies, as for example in the IPPC and Nitrates Directives.

Economic instruments (stimulation) are common in agricultural policy but are not very common in environmental policy (yet). Environmental taxes and tradable rights/quotas have only been implemented in a few countries. Subsidies are increasingly used as a policy instrument to promote environmentally friendly practices and the introduction of new technology.

Communicative instruments (persuasion) include collective projects to address environmental issues and measure to improve information flows to promote good practices and environmental objectives. This information can be provided to both producers, in the form of technical assistance and extension, and to consumers, via labelling. Technical assistance and extension are policy measures providing users with information and technical assistance to plan and implement environmentally friendly practices.

3.6 Governmental policies and measures in practice

Governmental legislation is a most important determinant for proper manure management and thus for reducing the environmental pressure arising from intensive livestock production.

Governmental legislations may contribute to overcome the following barriers for implementing proper manure management in practice:

1. Manure is considered to be a waste product, especially in the case of landless livestock production systems, which has to be disposed off at minimal economic costs.
2. Proper manure management is associated with considerable investments in capital, labor and knowledge, which increase the production costs.
3. The value of nutrients lost is relatively low compared to the costs associated with proper management and farmers therefore do not have any personal incentive to reduce emissions (losses).
4. Livestock farmers that do produce part of the animal feed often have little incentive to use slurry properly instead of fertilizers, because slurry is voluminous and has a low and variable nutrient content, while artificial fertilizer has a high and known nutrient content and is relatively cheap in developed countries.
5. The awareness and knowledge of farmers about the long-term consequences of environmental pollution through animal manure is still limited.

Though legislation may contribute to even out some of these barriers, there often is a gap between the theory and practice of governmental policy measures (Romstad *et al*., 1997; Gunningham and Grabosky, 1998). The variable and slow responses of environmental policies in agriculture have been ascribed to (i) the huge differences in farming systems and environmental conditions in practice, (ii) a variable interpretation of the governmental policy measures, (iii) legislative delays and lack of enforcement, (iv) failure by farmers to implement measures, due to within system constraints, perceived costs and the needed learning time, and (v) antagonisms between measures, due to lack of integration of measures. Moreover, there is lack of agreement about environmental protection at global level, at the level of the World Trade Organization. Within a globalizing world agreements about trade liberation and environmental protection must go hand in hand. Implementation of environmental policies at national level or regional levels creates unfair competition as intensive livestock production systems increasingly produce for the global market. Unfair competition is often a major reason why farmers in developed countries are reluctant to invest in proper manure management.

The EU Nitrates Directive (EC 1991) sets a limit of 170 kg per ha per year of manure N to be applied to agricultural land. This Directive is meant to reduce and prevent the pollution of groundwater and surface waters with nitrate from agricultural sources. Through its manure

application limit, it couples animal production to crop production indirectly within regions. This Directive has been heavily debated and has a significant influence on the further development of the intensive livestock production sector in the EU (Romstad *et al*., 1997). The Directive also includes various good agricultural practice guidelines for the proper application of manure and fertilizers to land.

Summarizing, environmental policy is a relatively new subject that emerged firstly in the European Union in the late 1980s. The policy measures aimed at decreasing N and P emissions to air and water. They have been developed and implemented in practice, while the understanding of the processes, cycling and functioning of N and P in the biosphere, in air, water, soil, plants and animals were still limited. In part because of our limited understanding of the complex N and P cycles, and because of the compartmentalization of our society and especially also governments, the focus in developing policy measures aimed has been on single N species (NO_3^-, NH_3, N_2O or NO_X), on single P measures, on single sectors (households, industries, traffic, crop production, animal production), on single environmental compartments (air, water, nature, humans), and or on single effects (human health, animal health, food quality, eutrophication, acidication, biodiversity loss, global warming). This multi-aspect approach has contributed to a ‘wealth’ of policies, with some interactive effects (both synergistic and antagonistic effects). As a result, there is an increasing quest for integrating environmental policy measures. Livestock farmers in especially EU feel squeezed by the large number of regulations. Yet, these regulations have contributed to improving the environmental performance of intensive livestock farms.

4. Outlook

Projections for 2030 suggest further increases in animal numbers in the range of 30 to 50 % relative to 2000, with largest increases for poultry and pigs kept in intensive livestock farms (Bruinsma, 2003). Increases will be relatively large in the developing countries near markets (large cities), while a (further) decrease is anticipated in some affluent countries, in response to globalization of markets, societal concerns about animal welfare and eating too much animal protein, and environmental policy measures. If there is no change in current practices, the projected increases in intensive livestock production systems will double the current environmental burden of livestock production, unless appropriate measures are taken.

Current trends of globalization and liberalization of agricultural markets, and the economics of scale, specialization and intensification will further contribute to the agglomeration of intensive

livestock production systems near rapidly developing markets. Small, mixed farming systems will be outcompeted and less people will be living in the country side. Some people in the country side may even eat animal products produced in urban areas, because it may be cheaper to produce meat in large facilities in urban areas than in small mixed farms in the countryside. This may further increase the environmental pressures of intensively livestock production systems, unless appropriate measures are taken.

These sobering projections are largely based on extrapolations of current trends. At the same time, there is overwhelming evidence that the environmental performances of livestock production systems can be improved greatly, when proper incentives are provided to livestock farmers, together with training, demonstrations, best available techniques and management tools. Currently, there is a lack of proper incentives, standards, and control in various areas with intensive livestock production systems. Clearly, there is a need of international agreements about animal production and animal manure management standards among countries with intensive livestock production systems. Such agreements should provide a basis for environmentally sound and socially acceptable production systems, and should also provide a level-playing ground to prevent unfair competition. The agreements should include regional livestock stocking density limits, expressed in livestock units per ha or preferably in P excretion or P surplus per ha.

Intensive livestock production systems can make a significant contribution to reducing the load of N and P into the wider environment. This requires cooperation among countries in setting standards for livestock production and manure management. It also requires collaboration among animal scientists, agronomists, technologists, economists and ecologists to derive these standards, and the required best available techniques and management tools. Foremost, it requires cooperation among the stakeholders along the food chain, including consumers, retail, food processing industry, representatives of livestock farmers, animal feed processing companies and technology companies. Adoption of standards and improved technologies requires additional skills, labor and investments in new technology and equipment by farmers. They need convincing information on expected costs and benefits. Evidently, a joined effort is needed.

Capacity building on good manure management means that reliable recommendations are available and accepted. To provide these is a challenge for joint activities of scientists, extension officers and progressive farmers and should also rely on pilot farms demonstrating the feasibility of the proposed measures.

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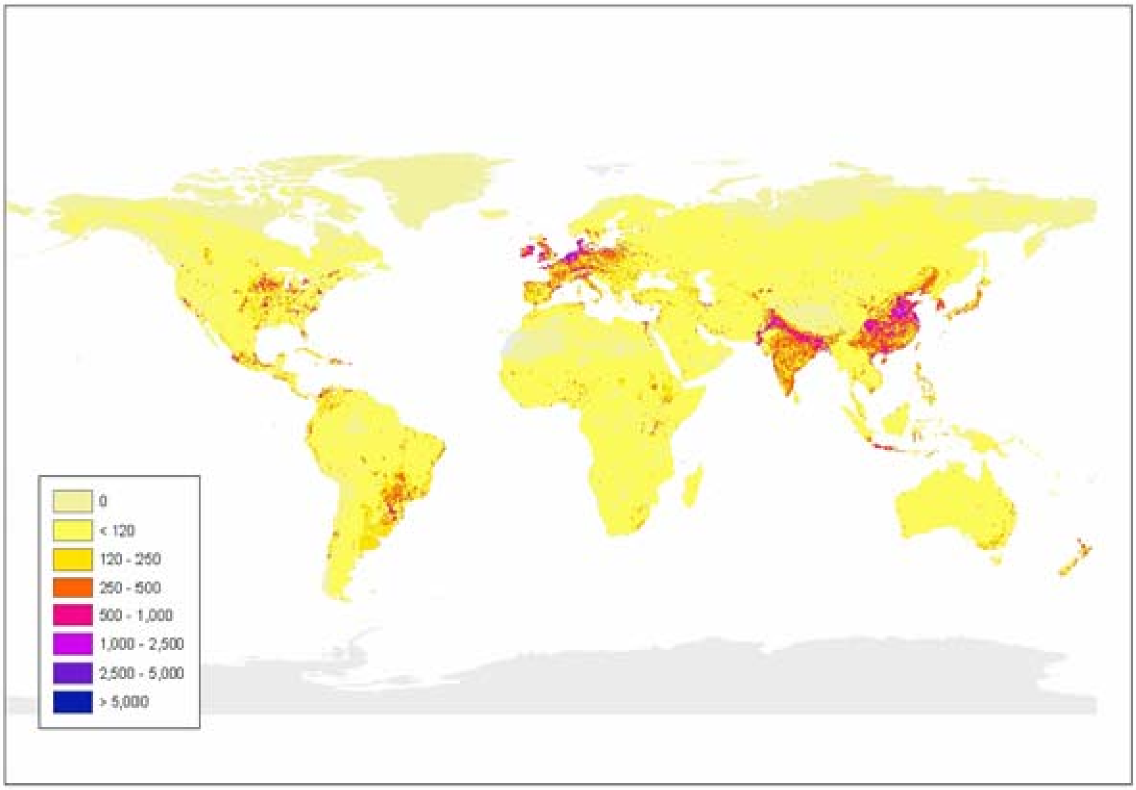

Fig. 1. Global distribution of the total amount of P_2O_5 in livestock excreta (kg km^{-2}), after Menzi *et al*., 2009.

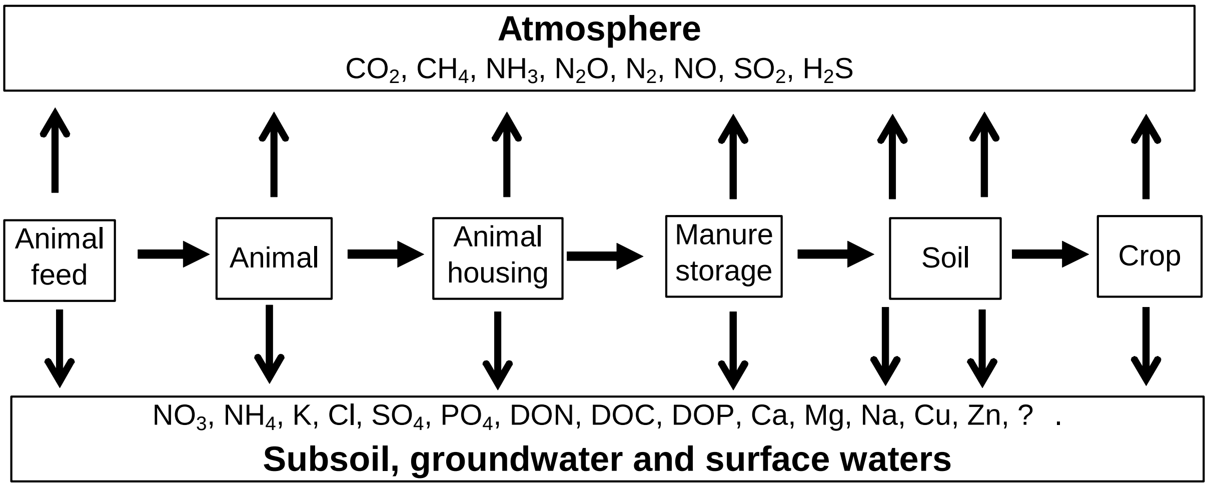

Fig. 2. Possible loss pathways of nutrient elements from the feed – animal – manure – soil – crop chain. Losses via gaseous emissions to the atmosphere are shown in the upper half, losses via leaching and run-off of nutrient elements to the subsoil and to groundwater and surface waters in the lower half (from Oenema *et al*., 2006)

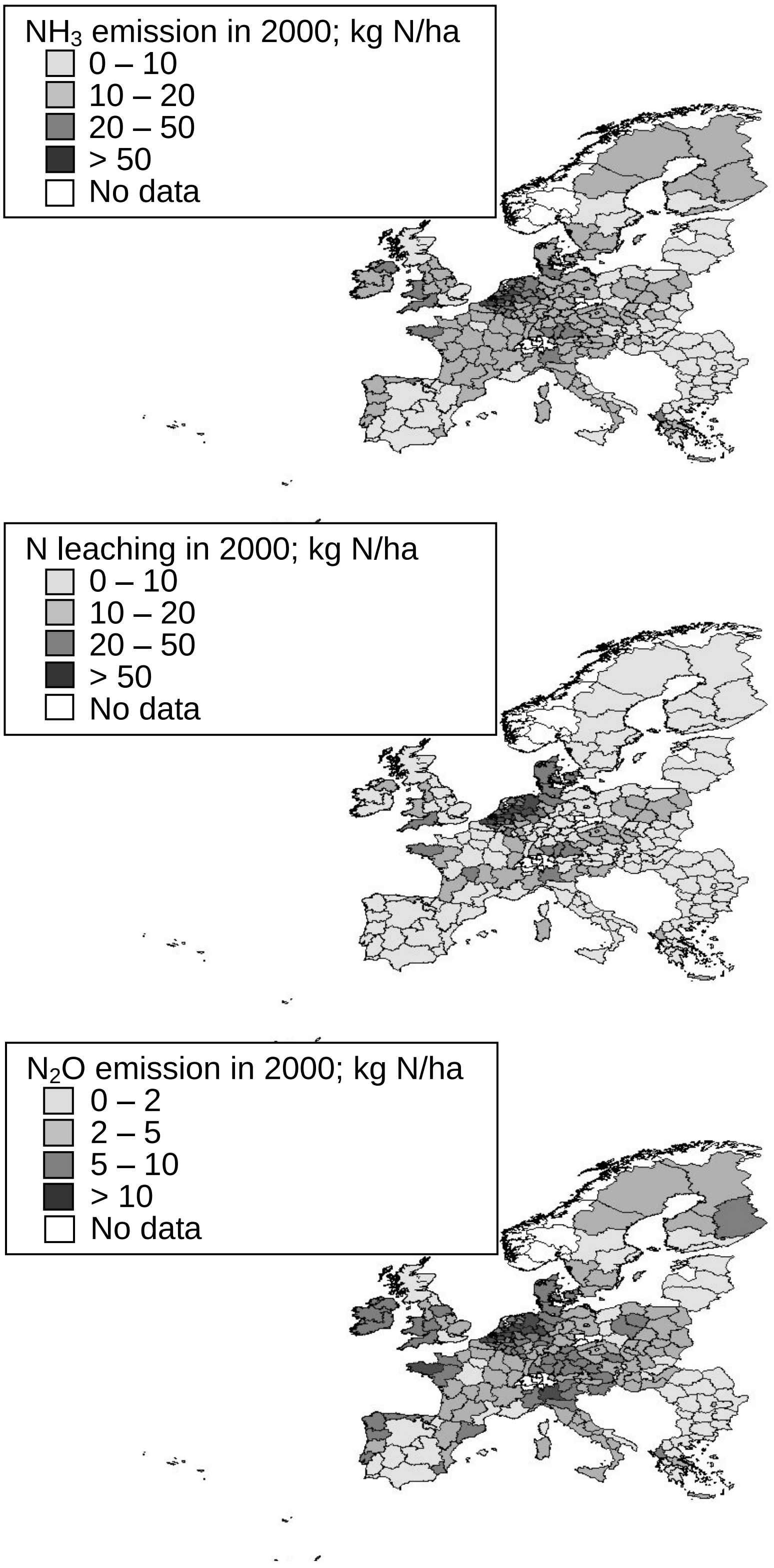

Fig. 3. Spatial distribution of ammonia emissions, nitrate leaching and nitrous oxide emissions in EU-27 (From Oenema *et al*., 2007).

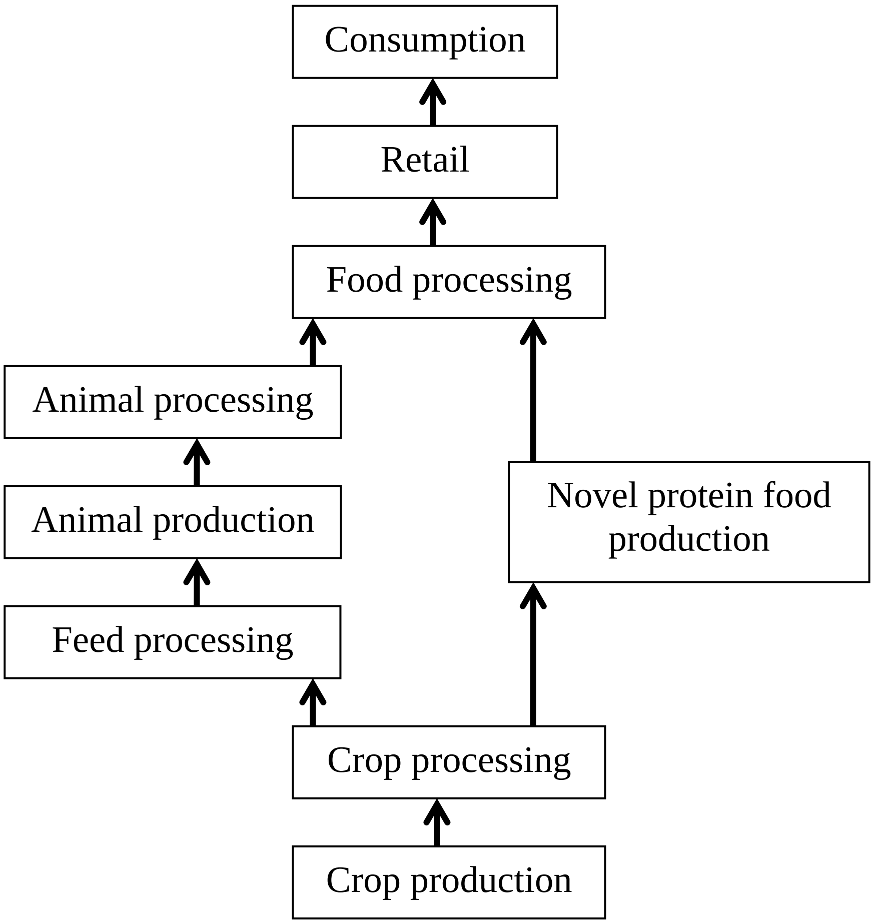

Fig. 4. The food chain from production to consumption. On the left-hand side the chain from crop production via animal production to consumption. On the right-hand side the chain from crop production via novel protein production to consumption. After Aiking *et al*., 2006.

Non-point Pollution of Nitrogen and Phosphorus from Agriculture in Taiwan

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Abstract

Reservoirs are the most important drinking water sources in Taiwan, however, most were eutrophicated due to nutrient overenrichment from watershed nonpoint sources (NPS). Except for the undistributed forest which occupied the largest area of the reservoir watershed, agriculture activities are the most important source of NPS pollution. Based on the NPS monitoring program enabled by Environmental Protection Administration and Water Resources Agency since 1995, data from rural catchments were selected to estimate the event mean concentrations (EMCs) and unit NPS loads. Based on those single-event data, the annual NPS loads for those studied catchments were derived by appropriate models to evaluate the NPS pollution from agriculture in Taiwan. The results indicated that the nitrogen and phosphorus loads from free-ranged chicken farm (FRCF) were almost 10 times to those from other rural catchments. Except for the FRCF, the average annual loads for recreation fruit farm, high mountain tea farm, betel palm farm, wasabi farm and broomcorn farm respectively are 5.1, 24, 8.3, 3.3 and 5.3 kg total nitrogen per hectare; 1.0, 0.24, 1.2, 6.7 and 0.96 kg total phosphorus per hectare.

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台灣農業非點源污染

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摘要

台灣飲用水水源有三分之二來自水庫，水庫水質影響民眾健康很大，而水庫水質優養造成藻類過多之繁殖，不但增加原水有機碳含量和三鹵甲烷潛勢，也促使毒藻的過量生長。如何減少水庫優養保護飲用水安全,是當今環保界最重要之課題。台灣水庫營養鹽的來源有 90%以上來自非點源污染，除了林地外，農業活動是最重要的來源。本文主要目的是把水庫集水區較重要的農業非點源污染調查結果加以統計並以模式分析，提供茶園、山葵園、檳榔園、放山雞養雞場、休閒農場和高粱與小麥輪種農田等非點源污染輸出係數，供從事農業經營者參考。研究顯示放山雞養雞場氮、磷輸出負荷很高，約為一般農地 10 倍以上；除養雞外，全年單位面積輸出 TN 負荷為高山茶園 (平均 24 kg/ha)、檳榔園 (平均 8.3 kg/ha)、高粱田 (5.3 kg/ha)、休閒農場 (5.1 kg/ha)及山葵園 (3.3 kg/ha)；TP 部分則依序為山葵園 (6.7 kg/ha)、檳榔園 (平均 1.2 kg/ha)、休閒農場 (1.0 kg/ha)、高粱田 (0.96 kg/ha)及高山茶園 (平均 0.24 kg/ha)。

關鍵詞：農業非點源污染，單位面積輸出之污染量，輸出係數，茶園，山葵園，檳榔園，高粱小麥輪作農田，放山雞養雞場

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前言

河川與水庫的水質與承受的污染量成正比例，污染分別來自外來和內部自生，其中又以外來的污染占主要部分。外來污染又可分成點源 (point source) 和非點源 (Non-point source, NPS)。最主要的點污染源是家庭污水、工業廢水及養豬廢水，而 NPS 主要是由降雨逕流，淋洗溶解地面和土壤中之物質所造成。以前自然界產生的逕流，被視爲背景逕流，並不認爲是污染，但近代人類活動頻繁，舉凡伐木建設、農業活動、工礦、車輛及生活廢污排放等產生之污染物，堆積在排水渠或土壤上，下雨將污染物沖洗而進入承受水體，造成污染，尤其久旱第一次暴雨或每場降雨前幾十分鐘所排的水，常含有可觀的污染物 (Chang *et al.*, 2008b)。NPS 污染主要來源有林區、農地、礦區、工地、建築區、市區、遊憩區、廢污處理場、河床淤泥等，主要污染物包括沉澱物、BOD、營養鹽、有毒物質、農藥、細菌及鹽類等。來自農業之 NPS 污染物，主要爲懸浮固體物（Suspended Solids, SS）、肥料中的氮、磷和農藥。

台灣河川中下游污染主要來自點源，而上游的污染主要來自 NPS，尤其水庫大約有百分之八十以上的污染來自 NPS，而台灣飲用水又有三分之二來自水庫，NPS 將造成水庫優養而導致後續飲用水處理的問題，水庫集水區除林地 (Chang *et al.*, 2008a) 外，農業是最重要之污染源。因 NPS 污染量測定非常困難，且模式推估仍需正確的實測資料率定與驗證，所以國內對農業流出的 NPS 污染資料非常少。李公哲等 （1987）在三峽農業地區測出晴天 COD, SS, TKN 和 TP 的單位污染量（輸出係數）分別爲 1.3, 8.1, 3.6 和 1.7 kg/ha/yr，至於全年之輸出係數則缺乏資料。中華水土保持學會（1996），根據國內外文獻資料及 81 年至 84 年在德基水庫集水區之土壤調查資料（楊秋忠，1995) 推估來自農果園和菜園 TP 的單位污染量分別爲 7.3 和 5.4 kg/ha/yr。日本小島貞男 (1984) 推估翡翠水庫集水區 TN 和 TP 之輸出係數分別爲 2.5 和 0.3 kg/ha/yr。國內輸出係數多爲推估值，仍需較多之現場調查資料才會可靠。

本文僅將十幾年來，環保署（溫清光等，1995，1996，1997，1998，1999，2000，2008）支持之 NPS 污染研究，及水利署南區水資源局（溫清光等，2001，2002）支持的水庫優養控制規劃，所測定之農業 NPS 污染量，做一統計分析，供大家參考。

材料與方法

1. 調查地點

本研究農地 NPS 污染調查，地點涵蓋台灣南部及金門地區之飲用水水庫集水區（如圖

1），包含檳榔、山葵園、高山茶園、休閒農場、放山雞養雞場及高粱田。各調查區的地形及基本特性列於表 1。

調查地點 1 (走馬瀨農場) 位於曾文溪中游，在自來水水源水質保護區內，下游有山上自來水場取水口，以休閒農場經營，面積 120 公頃，每年露營、採果遊客達 40 萬人次以上，污水收集經處理後排入曾文溪，但牧場、果園、遊樂場等之雨水逕流流入上潭和下潭後注入曾文溪，構成重要之 NPS 污染；調查地點 2~6 位於曾文水庫集水區內，曾文水庫爲台灣蓄水量最大及南部最重要的水庫，容量爲 54080 萬立方公尺，除了是國內最大農產地嘉南平原的主要灌溉水源外，水庫同時也供發電、工業用水及飲用水。根據近三年調查，水庫水質爲普養，但相當接近優養的臨界值 (卡爾森優養指標爲 45~50 之間)，水庫 75 %的營養鹽來自集水區的 NPS 污染 (Chang *et al*., 2004)。集水區主要土地利用爲天然林地，佔 70 %以上，農地使用最多爲檳榔（兩千多公頃）、其次依序爲茶園（一千多公頃且持續擴大）、果園及山葵園，本研究除了果園外，其餘都納入調查。茶及山葵都是高價值作物，因重施肥產生之 NPS 污染很可觀，爲瞭解不同種植方式對 NPS 負荷的影響，茶園有等高（編號 2，石桌茶園）及平台階段種植（編號 3，隙頂茶園）兩種，茶葉品種以清心烏龍及金萱爲主；檳榔挑選坡地（編號 4，永興檳榔園）與平地種植（編號 5，大埔檳榔園）兩類；山葵園選在種植密度最高的阿里山區（編號 6，阿里山山葵園），海拔爲 2300 公尺；調查地點 7、8 爲放山雞養雞場，分別位在二仁溪及曾文溪上游，養雞數分別約爲 10000 隻（編號 7，龍崎養雞場）及 5000 隻（編號 8，南化養雞場），養雞一年兩收，本研究採樣均在有養雞的週期內進行。金門農業以高粱與小麥輪作爲主，調查地點 9（斗門溪高粱田）屬於金沙溪流域，下游爲金沙水庫。金沙溪流域爲金門最大的高粱產地，砂質土壤，肥料極易流失，每年降雨淋洗出大量氮、磷，流至金沙水庫，使水庫常年處於超優養狀態（卡爾森優養指標爲 65~70 之間）。

2. 農業 NPS 污染負荷測定的方法

(1) 自動採樣設備

NPS 污染是由降雨逕流淋洗地表所引起，所以必須同時測定降雨過程的流量和污染物的濃度，爲測定大小不同的降雨事件以建立代表性 NPS 模式，污染量準確量測非常重要。台灣降雨特性不易掌握，加上水庫集水區都位於山區，測流量和採水樣非常困難，必須以自動採樣設備才能得到完整的數據。本文 1996 年測定走馬瀨休閒農場時，因經費限制自動化設備不足，採用人工採水樣和看尖頂堰的水位量流量，能測到之降雨場數較少。其他的農地皆採用

美國 ISCO 公司 6700 和 6712 型之自動採樣器，該型附有雨量計（674 Rain Gauge）和超聲波水位計（710 Ultrasonic Module）。ISCO 原設計自動採樣器的電池是 12V 電壓，電容小待機時間只能維持四天，爲了適應野外持時間待機，本計畫將原電池改裝，用汽車蓄電池代替，可待機 60 天左右。

(2) 暴雨期間水質自動採樣頻率

美國 ISCO 6700 或 6712 型的自動採樣器，可藉由降雨量或流量啓動採樣器、雨量計和水位計，依筆者之經驗，採用雨量啓動最適合。設定啓動條件，必須先研究調查區之降雨型態和可能持續時間，再應用機內之電腦軟體設定。如爲夏季降雨，常是強度大時間短，啓動條件降雨強度可設定 3mm/15min，採樣的頻率於短，若爲綿綿之春雨，需設定爲 2mm/30min。採樣頻率的設定，必須依據降雨的型態和集水區的面積，也就是及流時間的長短，因爲機器內裝之採樣瓶只有 24 支，夏季強度大持續時間短之暴雨，集水面積小的實驗場，尖峰流量在降雨開始後 5 至 10 分鐘就到達，所以最初幾瓶的採樣間距 3~5 min 採一次，尖峰流量過後可設定 10~20 min 採一次；若爲綿綿細雨季節，採樣間距需加長；颱風雨或梅雨下的時間長，中間必須拿空採樣瓶去替換裝滿水之採樣瓶回實驗室分析水質。分析項目有化學需氧量（COD）、總磷（TP）、正磷酸鹽（PO_4-P）、總凱氏氮（TKN）、氨氮(NH_3-N)、硝酸氮（NO_3-N）懸浮固體物（SS）等，分析方法依照環保署檢驗所公告或美國水質標準檢驗法（APHA, 1998）。

(3) 暴雨逕流測量設備和方法

超聲波水位計只能測水位，必須搭配尖頂堰，才可量流量。依集水區大小，估計最大流量，設計尖頂堰之形狀和尺寸，用合板或鍍鋅鋼板在現場依渠道形狀安裝。小集水區採用一個矩形堰（隙頂茶園、永興檳榔園、走馬瀨農場、南化及龍崎養雞場使用，以照片 1 爲例）或三角堰（阿里山山葵園、大埔檳榔園、石桌茶園，以照片 2 爲例），中集水區可採用梯形堰(永興檳榔園，如照片 3)，大集水區因流量變化大採用矩形複合堰（金門斗門溪，如照片 4），再配合超聲波水位計，每分鐘自動記錄堰頂水頭，儲存在記錄器內，再用各種尖頂堰之公式計算流量。台灣山區暴雨逕流之流量範圍大，很適合以複合堰量測，在降雨初期流量較小時，水位只達到第一個旁縮矩形堰，用矩形堰公式計算，如果流量較大，水位達到第二個矩形堰，分別計算兩個矩形堰之流量再相加。根據研究(Jan *et al.*, 2006)，複合堰採用兩個單矩形堰流量相加，其流量比實際小，誤差約 1~10 %（平均約 5 %），實際量測時這個誤差是可以接受的。

3. 農業 NPS 污染負荷之估計方法

台灣降雨時空分佈十分不均，加上很難取得足夠的 NPS 污染調查數據，因此，若只以採樣所得的數場降雨 (single-event) 推估污染負荷可能會產生很大的偏差，必須採統計或模式分析將實測數據進行推論以估計代表性的 NPS 污染量。NPS 污染估計方法有很多，包括簡易的單位係數輸出法、統計回歸法及模式分析法等，本研究採用模式法。

模擬 NPS 污染模式，包含降雨產生逕流及物質在地表或地面下產生和輸送的過程、因此包含逕流及水質的模擬，其模擬與水土保持界常見的輸砂模式類似。不過，NPS 污染模式包含營養鹽及物化水質之模擬，其模擬比土壤侵蝕複雜，再加上下雨時採樣及測定流量困難，不易取得大量的資料來率定和驗證模式，即使歐美已發展各種套裝模式，不過 NPS 負荷之模擬結果還是有很大的誤差。

爲減少模式適用性所造成的誤差，通常需視集水區空間尺度、數據時間尺度及使用者需求來選擇適當的模式。以尺度來區分，NPS 模式可分爲流域 (basin scale)、集水區 (watershed scale) 或小區域 (field scale) 模式；以時間尺度來分，可分爲單一暴雨 (AGNPS)、多場暴雨 (AnnAGNPS) 及連續性暴雨 (HSPF，SWMM) 模擬三種，單一暴雨一般用於評估平均 NPS 污染負荷，多場暴雨則同時考量乾期與雨期之污染流布情形，連續模擬可模擬短時間之連續性變化；若以用途來區分，可分爲簡易及管理分析模式二種，前者以實際數據之分析爲主，模式部份機轉以迴歸統計方程式或地區經驗式替代，優點是參數少且容易使用，在農業上最常使用的 AGNPS 模式 (李岳豐，1996) 即爲此類。管理分析型模式可供決策者依假想狀況進行模擬，牽涉到集水區各種作用與機轉，通常需要較多參數及較詳細的率定過程，因此又稱爲複雜模式，國內常用的有 HSPF 模式 (Liu *et al*., 2006; Chang *et al*., 2008a) 及 SWMM 模式 (Chang *et al*., 2008b)。

本研究現調資料有單場降雨短時間的降雨-逕流-污染負荷歷線，除了走馬瀨農場因降雨場次較少，採用參數少的 AGNPS 模式外，其餘可適用連續性暴雨模擬。在空間尺度方面，一般歐美發展的連續性暴雨模式都適用於中、大尺度集水區之模擬，因此本研究除了斗門溪高粱田採用 SWMM 模式 (溫清光，2008) 外，其他小集水區採用自行開發的小尺度降雨逕流-污染負荷模式進行模擬。有關 AGNPS 及 SWMM 模式之使用則請參照相關文獻，本文不再贅述。

本研究 7 處農業小集水區採用降雨逕流-污染負荷模式估計全年污染負荷，此模式屬小尺度複雜型連續降雨模擬，在逕流部分採用成大水利系游保杉教授發展之分佈型降雨逕流模式

(Yu and Jeng, 1997)，在污染負荷部分則根據各種懸浮及溶解性污染物經土壤侵蝕產生 (soil erosion)、溶解洗出 (washoff) 及泥沙釋出 (sediment yield) 等特性，選擇適用之污染負荷方程式 (loading functions)，結合降雨逕流模式計算出的流量模擬而得，以養雞場為例，整個模式架構如圖 2 所示。各場址的污染負荷方程式因地而異，詳細內容請參考相關文獻 (陳起鳳，2002；范楓旻，2002；李怡真，2003；吳致良，2004)。

結果與討論

1. 農業 NPS 污染測定結果

(1) 測定之天候條件

本研究 9 處採樣點的調查時間，有效調查場次及降雨概況列於表 2。台灣降雨集中在每年 5~8 月，主要可分為梅雨、山區午後雷陣雨、颱風及西南氣流造成的豪雨，場址調查均歷時一年以上，採得降雨多有涵蓋上述降雨類型，惟因調查年份及場址高程不同，各場址的降雨強度分佈也不相同，紀錄最大降雨量為 2001 年 9 月 18 日龍崎養雞場的 303 mm。列入計算之降雨事件均為可形成地表逕流之有效降雨，場次最少為走馬瀨農場之 2 場，場次最多為龍崎養雞場的 14 場，其餘均採得 5 場有效降雨以上。

(2) 農業活動雨水逕流污染物之濃度

農地雨水逕流濃度可反映集水區地形、地質、土壤、植披、耕作方式、作物類型及施肥型態，因一場降雨中不同時期的逕流濃度變異相當大，所以一般均採用降雨事件逕流平均濃度 (Event Mean Concentration, EMC) 來代表，EMC 由該場降雨總輸出污染負荷除以總逕流體積而得。一般來說，單場降雨初期逕流中之污染物濃度會比 EMC 高，末期則較 EMC 低 (梁銘基，2008)，這種現象在都市區比農業區明顯。同一地區，全年不同大小之降雨，每場次都有一個大小不同之 EMC 值，根據美國統計，這些 EMC 值大部分會成對數常態分佈，所以本研究所調查各場址之幾何平均 EMC 值列於表 3。兩處養雞場均位於陡坡，養雞密度高使場址內植披覆蓋程度低，再加上記錄到數場颱風及豪雨，所以 SS 濃度相當高已接近土石流水準；另大埔檳榔園及走馬瀨農場也因植披覆蓋程度低，所以 SS 較高；而平台種植的茶園 SS 明顯較等高種植者低。由 SS 的 EMC 可發現，各場址的 SS 主要受到降雨強度、植被、土壤性質及耕作方式的影響。

在有機污染方面，雞農通常只回收雞舍內之雞糞，戶外不回收的部分為放山雞養雞場主

要污染來源，因此有機性水質參數如 COD 及 TKN 特別高，造成相當嚴重的環境衛生問題；森林蔽蔭及雞糞打底使土壤有機質較高，使阿里山山葵園流出的 COD 濃度較其他農地高。

在無機鹽類方面，因 NO_3 在土壤中的移動力高，由其 EMC 可看出茶園氮肥的嚴重流失，研判兩處茶園的施氮肥量應遠高於建議值；除了養雞場流出 TN 以有機氮爲主之外，農地 NPS 污染的 TN 均以 NO_3 爲主要組成份。山葵及養雞場的磷 EMC 特別高，研判與其肥料及飼料中的高含磷量有關。此外，TP 與 SS 的相關性高，顯示有許多磷以泥沙爲載體而流出。

NPS 造成承受水體的影響主要來自泥沙之沖刷及無機鹽類，從台灣農地的的 EMC 分析結果看來，SS 雖主要爲天然因素所致，適當的坡度管理可有效減少沖刷；農地 NPS 流出的無機鹽類通常與施肥行爲有關，過量的氮肥會轉換爲高 NO_3-N 之流失；放山雞養雞場的飼養型態及山葵園的施肥習慣會造成來自於生糞便之有機污染。

(3) 農業活動之單位污染負荷測定值

測定之總污染負荷量與表面積及降雨量有關，本研究各調查地點之面積及降雨事件都不相同，因此採用單位降雨面積負荷 (g/ha-mm) 在同一基準下進行農地間流出 NPS 污染負荷之比較，結果如表 4，將集水面積及雨量正規化後，各場址輸出負荷的排名與 EMC 的結果仍然一致。透過單位降雨面積負荷可更清楚看出農地使用與 NPS 污染負荷的關係。除了養雞場之外，在磷的部分以高粱田及山葵園最高，TP 單位輸出負荷分別達 4.3 及 2.9 g/mm-ha；在氮的部分以高粱田及茶園較高，TN 單位輸出負荷分別爲 38.5 及 19 g/mm-ha。農地土壤中淋洗出的氮磷主要來自肥料，可合理推論當某種成分的營養素流失較多，表示該農地在正常產量下已超量供給該生長要素，造成污染。SS 輸出負荷與施肥及作物類型關聯性較低，主要與坡度及土壤有關，SS 輸出負荷較高者如大埔檳榔及高粱田屬砂質土；平台種植的茶園顯然對 NPS 污染負荷的控制比等高種植有效。

2. 農業活動之單位污染負荷模擬值

除了前述雨量與集水區面積之影響外，各種土地利用之 NPS 輸出負荷與還會與降雨強度、採樣代表性與降雨前污染物累積時間有關。研究指出，台灣水庫集水區的 NPS 污染負荷往往來自數場較大的降雨 (Chang *et al*., 2008a)。以金門高粱田及阿里山茶園爲例，高粱田的五場事件包含當年度最大的豪雨 (2007 年 8 月的聖帕颱風)，但阿里山茶園的採樣歷程中並沒有採到當年度最大的山區豪雨，所以吾人若直接以平均測定值來估計，可能會高估金門高粱田的負荷，但卻低估了阿里山茶園的真正負荷量。可見，用平均單位降雨面積負荷雖可消除雨量與集水區面積對輸出負荷的影響，不過若細究 NPS 的成因，我們還必需將場址調查時間、

降雨調查場次以及降雨規模等等條件都盡量一致後，才能進一步比較各農地間實際人爲影響的差異量。此外，國內常引用國外的文獻值做爲一般快速評估各種農地 NPS 污染負荷的簡易方法，由以上討論可發現，因國外氣候類型與農耕型態與國內差異很大，採用國外文獻係數推估本土污染一定會有更大的偏差。基於以上原因，國內有建置本土資料庫的必要，而資料庫中的數據還必須將各種影響因素一致化。

本研究利用測得數據率定模式參數，當模擬污染負荷可符合實測數據時 (以龍崎養雞場爲例，如圖 3)，再輸入全年參考降雨量，以此組率定後的參數模擬各農地代表水文年典型的 NPS 輸出負荷，計算結果如表 5。SS 負荷推演到全年後，與測定值相比有增加者可看出植被覆蓋 (休閒農場) 及坡度 (山葵、坡地檳榔) 對 NPS 的影響；等高與平台茶園的差距降低；因金門高粱田測定值以大雨爲主，推演至全年後排名降低；在營養鹽部分，高山茶園的 NO_3-N 及 TN 最高，即使以平台種植仍有很高的氮輸出，尤其是硝態氮；金門高粱田總磷輸出降低，使山葵園成爲特別突出的 PO_4-P 及 TP 農業區。兩個養雞場污染負荷與養雞隻數未成正比，因此用本文單位雞隻負荷與雞隻數估計其他放山雞養雞場會有偏差，無論如何放山雞養雞場氮磷輸出十分可觀，以一公頃 8000 隻雞估計，其輸出負荷爲一般農地 10 倍以上。

結論

本文將十幾年來筆者實際參與之國內農業區 NPS 調查背景、方法及結果做一整理，主要結論有下列幾點：

1. 農地類型及施肥行爲會影響降雨逕流水質和 NPS 污染負荷，茶樹以施氮肥爲主，加上無機氮溶解度高，所以茶園流出的雨水氮濃度與負荷較高；山葵和高粱施肥以磷肥較重，流出雨水以磷較嚴重。
2. 同一類型 (如茶園) 但耕作方式不同的農地，在 NPS 污染輸出負荷上會有顯著差異，平台種植可有效減少土壤與肥料流失。
3. 放山雞養雞場排出之 NPS 荷含高濃度有機質污染物，已接近廢水程度，若未經處理即逕行排放會造成嚴重的環境衛生問題。
4. 曾文水庫集水區在 TP 部分以山葵園單位面積負荷較高，TN 則以高山茶園最高。
5. 金門斗門溪高粱田屬砂地，肥料中的營養鹽無論氮磷均大量流失，使承受溪流長滿大量水生植物並造成水庫優養。

6. 除養雞場外，本研究調查農業區全年單位面積輸出 TN 負荷依序為高山茶園 (平均 24 kg/ha)、檳榔園 (平均 8.3 kg/ha)、高粱田 (5.3 kg/ha)、休閒農場 (5.1 kg/ha)及山葵園 (3.3 kg/ha)；在 TP 部分則依序為山葵園 (6.7 kg/ha)、檳榔園 (平均 1.2 kg/ha)、休閒農場 (1.0 kg/ha)、高粱田 (0.96 kg/ha)及高山茶園 (平均 0.24 kg/ha)。

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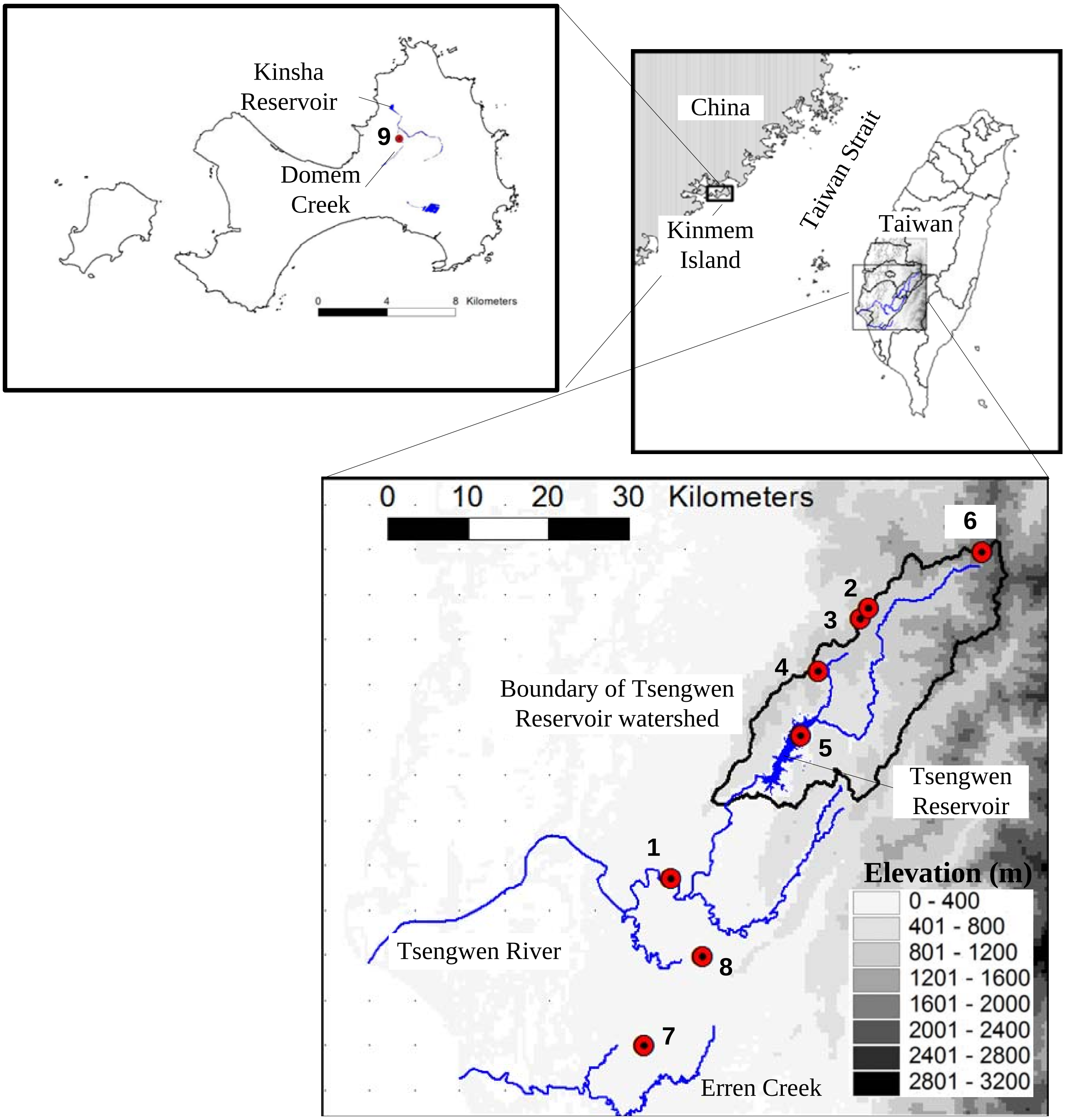

圖 1. 本研究農業非點源污染調查場址分佈圖

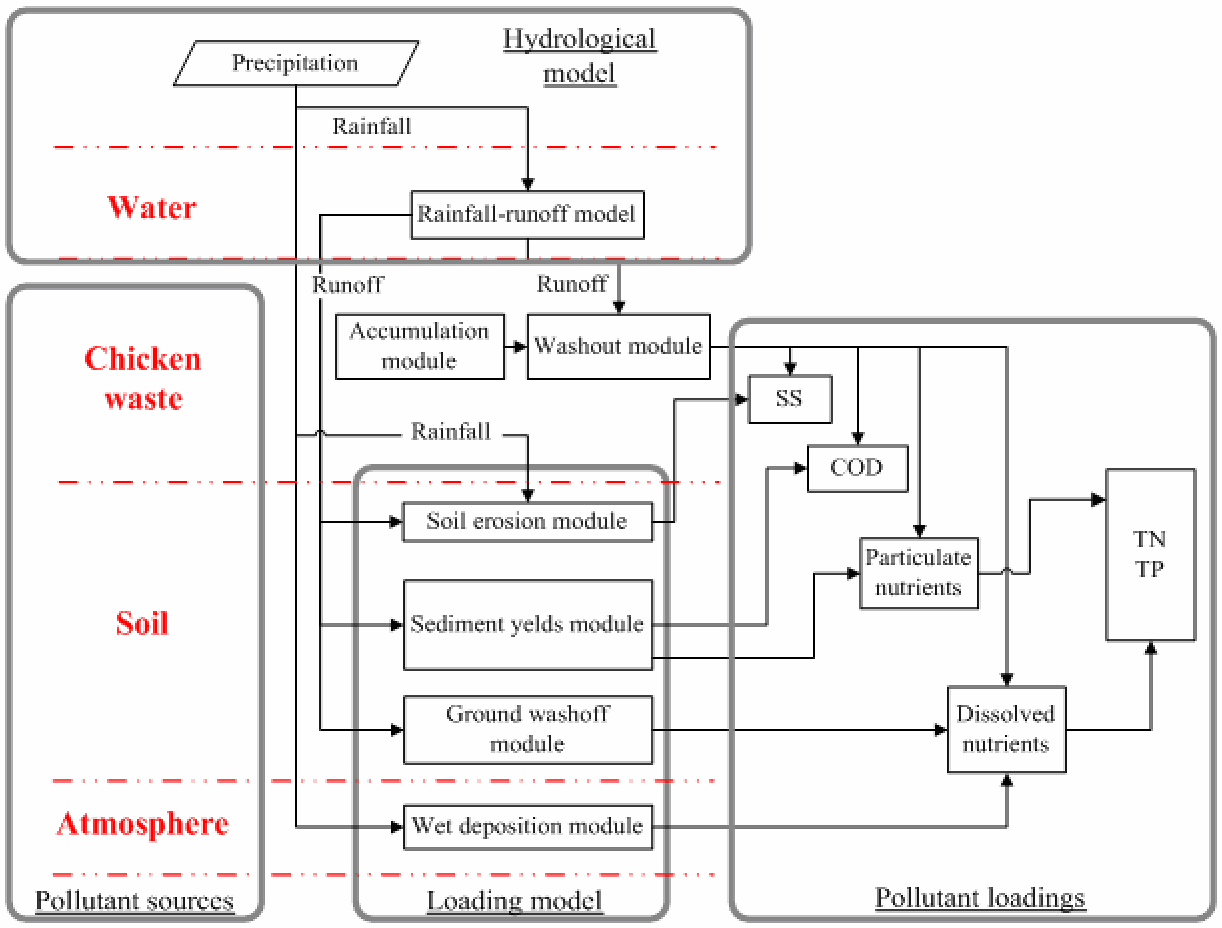

圖 2. 以放山雞養雞場爲例說明本研究非點源污染模式概念及流程圖

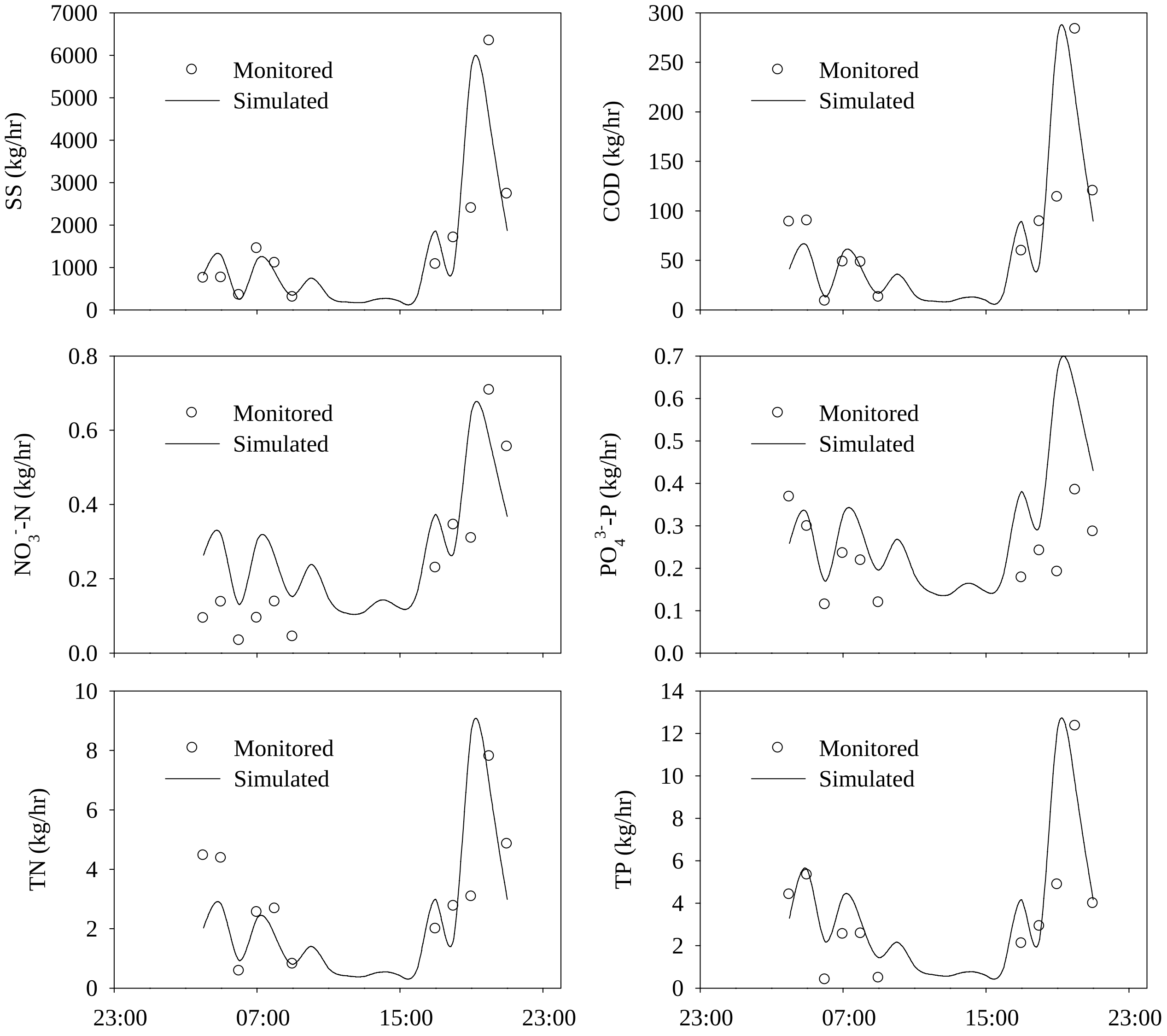

圖 3. 放山雞養雞場(C1)單場降雨非點源污染歷線及模擬結果之比較(以 2001 年 9 月 18 日降雨事件爲例)

表 1. 本研究調查農業區及其特徵

場址編號	場址代碼	植栽	類型	海拔 (m)	坡度 (%)	集水面積 (ha)
1	F1	果樹	休閒農場	200	5-40.0	31.0
2	Tea1	高山茶	等高種植	950	35.0	9.6
3	Tea2	高山茶	平台種植	900	30.0	1.3
4	Betel1	檳榔	斜坡種植	800	26.0	90.0
5	Betel2	檳榔	平地種植	320	2.0	0.5
6	W1	山葵	斜坡帶狀種植	2300	30.0	0.3
7	C1	放山雞	一萬隻放山雞養雞場	200	50.0	1.3
8	C2	放山雞	五千隻放山雞養雞場	400	15.0	0.6
9	B1	高粱小麥輪作	集區式種植	30	0.5	440.0

表 2. 本研究調查期間各場址之降雨狀況

場址代碼	調查期間	場次	降雨逕流深度 (m)	降雨量 (mm)	平均降雨強度 (mm/hr)
F1	Jul., 1995－Apr., 1996	2	0.02-0.34	20-50	7-13
Tea1	Apr., 2001－Sep., 2001	13	0-0.19	12-145	1-20
Tea2	Sep., 2001－Aug., 2002	10	0-0.31	12-239	2-19
Betel1	Jan., 2001－Jun., 2002	7	0.02-13.66	11-182	0-11
Betel2	Jun., 2002－Sep., 2003	13	0.01-0.25	14-217	2-16
W1	Jan., 2002－Aug., 2002	6	0-0.01	12-108	4-15
C1	Aug., 2001－Jul., 2002	14	0-1.17	8-303	1-40
C2	Jul., 2002－Aug., 2003	7	0-0.5	14-254	2-34
B1	Aug., 2007－Jul., 2008	5	6.3-138.5	26-116	2-4

表 3. 本研究調查農業區之平均事件降雨逕流濃度

場址代碼	SS	COD	TKN	NH3-N	NO3-N	TN	PO4-P	TP
F1	292	44	2.9	-	-	4.5	-	0.23
Tea1	164	77	1.5	0.4	13.1	14.5	0.11	0.50
Tea2	9	14	1.7	0.8	11.2	12.9	0.06	0.13
Betel1	57	10	1.3	0.5	1.6	2.9	0.05	0.17
Betel2	324	56	2.1	1.3	1.0	3.4	0.45	1.32
W1	476	214	0.6	0.1	0.9	1.5	3.00	4.50
C1	12978	2572	87.9	21.9	0.4	86.4	5.51	70.00
C2	7013	3993	169.0	138.0	0.3	268.0	22.00	111.00
B1	194	33	1.6	0.4	1.9	3.6	0.18	0.74

Unit: mg/L

表 4. 調查農業區之平均單位 NPS 流出負荷實測值

場址代碼	SS	COD	TKN	NH3-N	NO3-N	TN	PO4-P	TP
F1	138	18	1.6	-	0.02	-	-	0.04
Tea1	141	45	0.7	3.9	14.60	19.0	0.08	0.50
Tea2	2	4	0.5	0.2	6.20	6.7	0.93	1.90
Betel1	31	9	0.8	0.5	1.50	2.3	0.03	0.10
Betel2	1038	170	8.4	5.2	3.50	12.4	1.40	3.90
W1	311	126	2.9	0.1	0.60	1.0	2.00	2.90
C1	50914	10278	356.0	88.0	1.80	351.0	22.00	279.00
C2	29695	17102	1000.0	909.0	1.50	1682.0	105.00	598.00
B1	1100	209	9.5	5.7	28.20	38.5	1.16	4.32

Unit: g/ha-mm

表 **5.** 模式推估之農業區單位面積年 NPS 流出負荷

場址代碼	參考水文年降雨量 (mm/yr)	單位面積年 NPS 流出負荷 (kg/ha-yr)					
		SS	COD	NO3-N	TN	PO4-P	TP
F1	1264	1233	21	-	5.1	-	1.00
Tea1	2160	89	-	33.10	33.6	0.14	0.22
Tea2	2218	63	-	12.70	14.4	0.14	0.26
Betel1	2306	100	60	-	14.9	-	0.32
Betel2	1928	1497	-	-	1.6	1.07	2.12
W1	2197	597	-	1.10	3.3	4.50	6.70
C1	1660	810 a	419 a	0.02 a	1.3 a	1.33 a	4.67 a
C2	1212	5373 a	418 a	0.80 a	50.0 a	2.60 a	57.20 a
B1	1007	313	40	1.60	5.3	0.19	0.96

註 a: 放山雞養雞場 NPS 流出負荷已轉換爲每隻雞每年輸出負荷 (g/c-yr)

照片 1 束縮矩形堰

(場址代碼：T2)

照片 2 束縮式三角堰

(場址代碼：T1)

照片 3 梯型堰

(場址代碼：Betel1)

照片 2 複合堰

(場址代碼：B1)

Status of Irrigation Water Quality Polluted by Livestock Industry in Taiwan

Yu-Hsiueh Liu[1,2] and Ta-Wei Chang1[1]

Abstract

According to the statistic data of 2008, there are 2,272 livestock plants, which were quickly raised since 2003, discharged waste water into irrigation canals. A major part of those plants are located at Yun-Lin, Chung-Hwa, and Chia-Nan Irrigation Associations, respectively. In detail of the statistic data of the preliminary test (also called field analysis) shows that 89% of the discharged canals belong to drainage canal in which the water quality should be meet the standard of effluent water. However, there is no electrical conductivity (EC) item in effluent water standard. On the other hand, there are 11 % of the discharged canals belong to dual-purpose (irrigation and drainage) canal and recycled canal in which the water quality should be meet the standard of irrigation water quality, which include EC item. In addition, not only livestock plants run up promptly but also EC value have the tendency to increase year by year from 2003. There are 76 % to 80 % of the EC value ranged two criteria 1,501 ~ 2,250μS/cm to >3,000μS/cm. In contrast to preliminary test, in 2008 there are 405 double-checking analysis around Taiwan for more detailed chemical analysis. Unqualified items of the highest rate are EC (54 %) and Ammonia (47 %) and the unqualified rate of suspended solid, chloride, sulfate, copper, and zinc are 11.4 %, 14.8 %, 8.1 %, 3.5 %, and 3.0 %, respectively. In past 2 years, there are 21 suspected pollution farmland sites which were surveyed, sampled, and analyzed. Eleven sites of the EC of saturated soil extract solution are greater than 1,000μS/cm which is already reached the FAO criteria of effecting banana and beans growth. Among them there is one site that the EC of saturated soil extract solution is greater than 3,000μS/cm which may influence the growth of the rice. Meanwhile, there are 2 and 5 sites containing high copper (> 100 mg/kg) and zinc (> 200 mg/kg) content which is higher than other sites irrigated by non-pollution irrigation water. Accelerating accumulation of copper and zinc in crop will reaching the control criteria of heavy metal in edible crop (200 mg/kg and 600 mg/kg, respectively) if suffer this kind of polluted irrigation water again.

Key Words: Livestock industry, Irrigation water quality, Electrical conductivity, Heavy metal.

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台灣地區灌溉水質受畜牧業污染之現況

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摘要

97 年度 15 個水利會畜牧搭排戶數總計有 2,272 戶，主要集中於雲林、彰化及嘉南等水利會。依據 97 年度畜牧廢水所搭排渠道類別結果顯示，主要分布於排水路約佔 89 %，廢水是以放流水標準管制；另排入灌排兼用、迴歸利用渠道則以灌溉水質標準 EC 規範，兩類型搭排戶約佔總搭排戶之 11 %。

彙整90~96年全省水利會畜牧業搭排戶水質初驗資料，發現台灣地區搭排戶放流水EC值有逐年增加趨勢，主要介於1,501~＞3,000 μS/cm兩個等級，約佔76 %~80 %。

蒐集 92~96 年間水利會畜牧廢水影響之灌排渠道 405 筆水質複驗資料，比較各項不合格率，仍以電導度 54 %爲最高，其次爲氨氮(47 %)，另外 SS、氯鹽與硫酸鹽等項目均介於 10 %，另重金屬則為 Cu 3.5 %、Zn 2.9 %。

96~97 年度完成 21 處污染灌區農田土壤飽和液 EC 與重金屬 Cu 與 Zn 檢測，結果顯示，土壤 EC>1000 μS/cm 總計 11 處，其鹽份累積已達 FAO 之影響香蕉、豆類等範圍，其中有 1 處農田土壤>3000 μS/cm，其可能影響水稻之生長。農田土壤重金屬 Cu>100 mg/kg 有 2 處、Zn>200 mg/kg 有 5 處，各灌區農田土壤兩項重金屬含量均較引灌乾淨水之土壤高，因此若再遭受此類灌溉水污染，其農田土壤 Cu、Zn 將累計達食用作物管制標準 200 mg/kg、600 mg/kg。

關鍵詞: 電導度、氨氮、重金屬

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前言

台灣地區農田水利會灌區之主要污染源來自工業與畜牧業兩大類，但各縣市政府下水道普及率偏低下，兩類產業之廢水經常排入農田水利會之灌排水路而影響灌溉水質，至於畜牧業主要集中於彰化、雲林、嘉南、高雄與屏東等中南部水利會灌區，因畜牧場廢水排入水利會灌排水路，且農田水利會灌溉水量又不充足下，因而部份灌溉水路之水質遭受污染。

本文蒐集農田水利會與農委會委託本中心之專案計畫之資料，進行統計與評估，以探討畜牧業排放廢水排入各灌排渠道之影響，一般搭排戶 (包括畜牧業) 搭排申請是以灌溉水質標準管理與規範，但自 1990 年起因畜牧單位之要求，則放寬灌溉水質標準 EC 等項目之管制，因此在文中將就著本措施之施行對畜牧廢水與灌溉水質之影響進行比較與評析。畜牧污染灌區專案調查，進行其相關畜牧業廢水處理設施現勘、污染灌區農田土壤採樣進行飽和液電導度與重金屬銅與鋅總量之檢測，以了解遭受畜牧污染灌區農田土壤無機鹽與重金屬累聚情形。

材料與方法

本文針對台灣地區畜牧業搭排戶數與廢水量統計、畜牧業排放廢水之水質、遭受污染渠道水質、污染灌區畜牧業廢水處理設施現勘與灌區農田土壤 EC、重金屬 Cu 與 Zn 之採樣分析工作，進行畜牧業廢水對農業環境影響之檢討與評估，各工作項之方法如下所述:

一、歷年畜牧業搭排戶數、搭排水量等統計

由行政院農委會「農田水利會灌溉水質監測管理系統」蒐集畜牧業廢水相關基本資料並進行廢水量等統計，以臺灣省 15 個水利會同意搭排申請者爲主，依水利會別、管理處別、工作站別、渠道類別(區分爲灌溉專用、灌排兼用、迴歸利用、排水專用等四類)予以統計與檢討。

二、歷年 (90 年~96 年) 畜牧業排放廢水 EC 值變化趨勢

包括蒐集農委會台灣地區養豬頭數與戶數、畜牧業搭排戶數統計，並比較歷年(90 年~96 年)畜牧排放廢水 EC 值變化趨勢，畜牧業放流水質枯豐水期水質之比較。

三、畜牧廢水影響灌排渠道水質現況

統計歷年 (89 年~96 年) 遭受畜牧廢水搭排渠道水質初驗 EC，並評估歷年遭受畜牧廢水搭排渠道水質複驗資料，以探討全省水利會灌排渠道水質污染現況。

四、畜牧污染灌區灌排渠道灌區農田土壤電導度 (EC)、重金屬等檢測

分別在 96~97 年期間採取 21 處污染灌排渠道灌區之農田土壤，採樣方法為採取進水口 3 公尺內表土 5 處混樣，並攜回實驗室自然風乾後，進行土壤飽和液 EC 值檢驗，並分析土壤重金屬 Cu 與 Zn 總量濃度，同時在各灌區也採取以地下水灌溉之農田土壤進行三項之檢測，探討兩類灌溉水質對農田土壤無機鹽與重金屬之累積情形。

結果與討論

一、歷年畜牧業搭排戶數、搭排水量等統計

比較歷年 (89~96 年) 搭排戶數與年搭排廢水量，均呈現逐年升高之趨勢，累計至 96 年全省水利會搭排戶數計有 2,361 戶，年搭排廢水量為 3,134 萬噸/年 (如圖 1~圖 2 所示)。搭排戶主要集中於雲林、彰化及嘉南水利會，分別佔所有搭排戶數 1,132 (48 %) 戶、418 (18 %) 戶、375 (16 %) 戶，其次為高雄水利會 134 (5.7 %) 戶及屏東水利會 128 (5.4 %) 戶；年搭排廢水量則以彰化水利會為最大 (1,571 萬噸/年)、其次為雲林水利會 (1,020 萬噸/年)、另為嘉南水利會 (208 萬噸/年)。由數據顯示，全省之畜牧業搭排戶自 92 年開始迅速增加。

依據 97 年度畜牧廢水所搭排渠道類別統計，結果顯示，其主要分布於排水路等約有 2,014 戶 (佔 88.6 %)，廢水排在此類型水路是以放流水標準管制之，並無電導度 (EC) 規範；另排在灌排兼用、迴歸利用渠道才以灌溉水質標準 EC 規範之，兩類型搭排戶分別有 152 戶 (佔 6.7 %)、106 戶佔 (4.7 %)，總計有 258 戶約佔總搭排戶之 11.4 %。其主要在雲林、彰化與高雄與屏東等地區。

另依據行政院農業委員會 96 年 11 月底公告統計之養豬統計資料，台灣地區畜牧業者共計 11,705 戶，但 97 年度農田水利處統計在各水利會灌區有申請搭排之畜牧業者計 2,272 戶，其搭排率佔總飼養戶 19.4 %。搭排戶僅 258 戶廢水排入灌溉渠道 (灌排兼用與迴歸使用)，其約佔台灣地區總飼養戶 (11,705 戶) 之 2.2 %，此類搭排戶才需以灌溉水質標準管制。

二、歷年 (90 年~96 年) 畜牧業排放廢水 EC 值變化趨勢

彙整 90 年至 96 年全省水利會畜牧業搭排戶水質初驗資料，以灌溉用水水質電導度 (EC) 項進行分析，將歷年 EC 監測資料依≦750 μS/cm、751~1,500 μS/cm、1,501~2,250

μS/cm、2,251~3000 μS/cm、＞3,000 μS/cm 等級區分，並繪製各年度在各等級之百分比，如圖 4 所示。由圖中顯示 90 年 EC 符合灌溉用水水質標準≦750 μS/cm 之畜牧搭排戶比例 (23 %) 明顯較高，至 96 年畜牧業搭排戶放流水水質符合灌溉用水水質標準僅佔 12 %，顯然降低 11.0 %，另介於 751~1,500 μS/cm 者亦較 90 年亦減少 9.8 %。

另比較歷年 (90 年至 96 年) EC 在各等級百分比之變化趨勢，以 1,501~2,250 μS/cm 等級增加 8.2 %為最多，其次為 2,251~3000 μS/cm 等級增加 5.4 %，另＞3,000 μS/cm 等級增加 7.2 %，由不合格率分析可以歸納出，自 89 年畜牧業廢水暫緩檢測 EC 後，台灣地區搭排戶放流水 EC 有逐年增加之趨勢，主要介於 1,501~2,250 μS/cm 及＞3,000 μS/cm 兩個等級，約佔 76 %~80 %。

三、畜牧廢水影響灌排渠道水質現況

彙整 89 年~96 年遭受畜牧廢水影響之渠道水質資料 7,425 點次初驗水質資料 (表 1)，並比較由 89 年起放寬 EC 管制下，各灌排渠道水質是否有遭受影響而劣化，表 1 為各監測點在各年度 EC 之平均值，資料顯示 96 年度灌排水質 EC 較 89 年 (或 90 年與 91 年) 增加地區，包括北基 (+32 μS/cm)、苗栗 (+286 μS/cm)、台中 (+106 μS/cm)、彰化 (+14 μS/cm)、屏東 (+259 μS/cm)、台東 (+25 μS/cm)、花蓮 (+3 μS/cm)，增加趨勢最多則以苗栗與屏東兩水利會。

以 96 年監測資料可見彰化、雲林、高雄與屏東統計值均超過灌溉用水水質標準之。但宜蘭、北基、苗栗、台中、南投、台東及花蓮等地區因 EC 多數<750 μS/cm，故屬暫無明顯影響；桃園地區 EC<1500 μS/cm 者為多數，為部分地區有影響者；彰化、雲林、嘉南、高雄及屏東等地區 EC 多數介於 751~1500 μS/cm 與>1501 μS/cm，影響區位面廣，故屬部份地區有嚴重影響。

蒐集 92~96 年度期間畜牧廢污水排入之灌排渠道之水質複驗資料總計有 405 點次 (表 2)，各項檢驗結果以不合格率，仍以電導度 54.3 %為最高，其次為氨氮 (46.9 %) 該項，另外 SS、氯鹽與硫酸鹽等項均介於 10 %，另重金屬則為 Cu 3.5 %、Zn 2.9 %，但這些地區部份灌排水路仍有其他類別產業廢水匯入，要界定其屬於畜牧污染仍有其困難。

四、畜牧污染灌區灌排渠道灌區農田土壤電導度 (EC)、重金屬等檢測

96~97 年完成 21 處畜牧污染灌區專案調查，結果發現有 7 處 (33.3 %)灌溉水質之 EC 有增加惡化之趨勢，現勘其所屬畜牧業有 87 戶，並比較 97 年與歷年之排放水，其排放水質 EC 昇高者有 46 戶 (53 %)，顯然其排放水在無 EC 管制下，部份排放水質 EC 有增加與惡化現象。

另現勘灌區畜牧業廢水處理設施，結果顯示，有處理設施有 83 戶 (97 %)，但有實際操作業者有 57 戶 (66 %)，但部份處理設施損壞，尤其厭氣段之紅泥塑膠袋因長年在日光暴晒下，破損無法發揮其功效，另部份業者廢水處理池並無廢水進入而呈乾涸狀態，仍需主管單位加強輔導強化其處理設施。

兩年度進行污染灌區農田土壤飽和液 EC 與重金屬 Cu 與鋅之檢測，結果顯示，土壤 EC>2000 μS/cm 有 6 處、>1000 μS/cm 有 5 處，總計 11 處 (52 %) 鹽份累積已達 FAO 之影響香蕉、豆類等旱作範圍，其中有 1 處灌區農田土壤>3000 μS/cm，其可能造成水稻生長之影響。農田土壤重金屬 Cu>100 mg/kg 有 2 處、Zn>200 mg/kg 有 5 處，各灌區農田土壤兩項重金屬含量均較引灌乾淨水之土壤高，因此若再遭受此類灌溉水之污染，其農田土壤 Cu、Zn 將累計達食用作物管制標準 200 mg/kg、600 mg/kg，而造成農作物毒害並影響人畜食用之安全。

結論

1. 97 年度 15 個水利會畜牧搭排戶數總計有 2,272 戶，主要集中於雲林、彰化及嘉南等水利會。由數據顯示，全省之畜牧業搭排戶自 92 年開始迅速增加。
2. 依據97年度畜牧廢水所搭排渠道類別統計之，結果顯示，其主要分布於排水路等約有2,014戶 (佔89 %)，廢水排在此類型水路是以放流水標準管制之，並無電導度 (EC) 規範；另排在灌排兼用、迴歸利用渠道才以灌溉水質標準EC規範之，兩類型搭排戶總計有258戶約佔總搭排戶之11 %。其主要在雲林、彰化與高雄與屏東等地區。
3. 彙整90年至96年全省水利會畜牧業搭排戶水質初驗資料，以灌溉用水水質電導度 (EC) 項進行分析，自89年畜牧業廢水暫緩檢測EC後，台灣地區搭排戶放流水EC有逐年增加之趨勢，主要介於1,501~2,250 μS/cm及＞3,000 μS/cm兩個等級，約佔76.0 %~80.0 %。
4. 蒐集 92~96 年度期間水利會畜牧廢污水排入之灌排渠道之 405 筆水質複驗資料，比較各項之不合格率，仍以電導度 54 %為最高，其次為氨氮 (47 %) 該項，另外 SS、氯鹽硫酸鹽等項均介於 10 %，另重金屬則為 Cu 3.5 %、Zn 2.9 %。
5. 96~97 年度進行 21 處污染灌區農田土壤飽和液 EC 與重金屬 Cu 與鋅之檢測，結果顯示，土壤 EC>2000 μS/cm 有 6 處、>1000 μS/cm 有 5 處，總計 11 處 (52 %) 鹽份累積已達 FAO 之影響香蕉、豆類等旱作範圍，其中有 1 處灌區農田土壤>3000 μS/cm，其可能造成水稻生長之影響。農田土壤重金屬 Cu>100 mg/kg 有 2 處、Zn>200 mg/kg 有 5 處，各灌區農

田土壤兩項重金屬含量均較引灌乾淨水之土壤高，因此若再遭受此類灌溉水之污染，其農田土壤 Cu、Zn 將累計達食用作物管制標準 200 mg/kg、600 mg/kg，而造成農作物毒害並影響人畜食用之安全。

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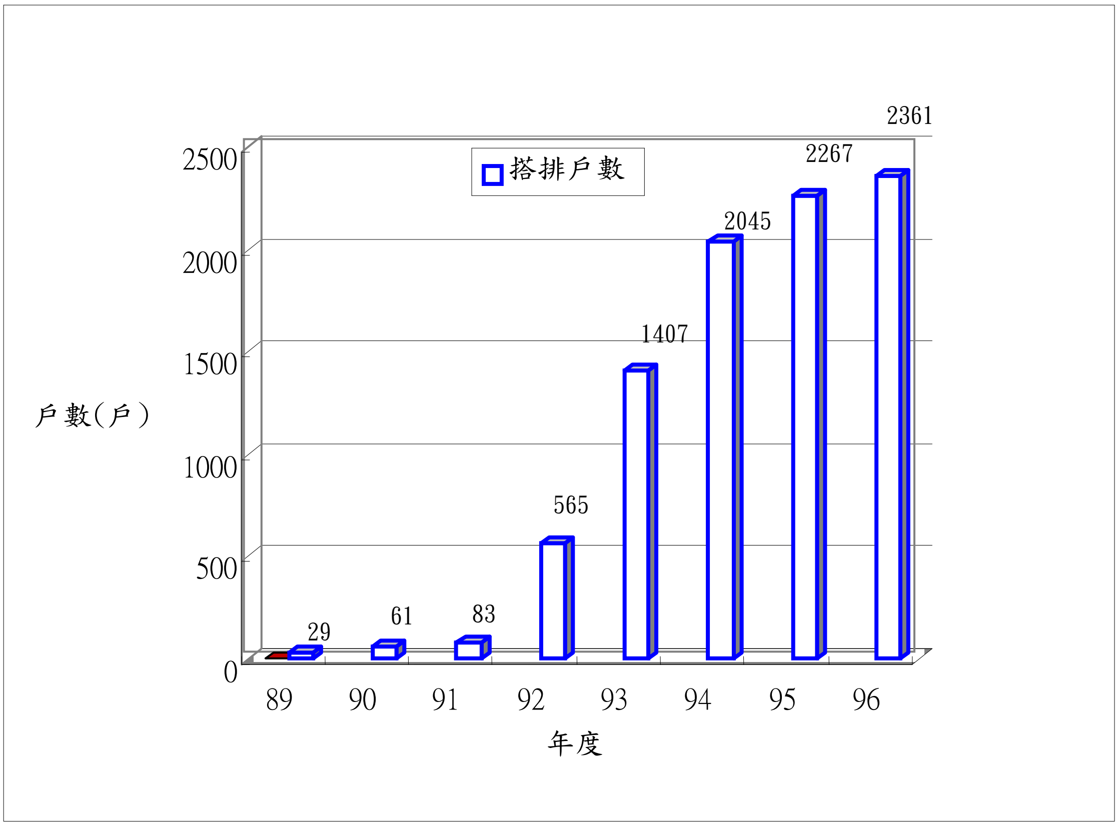

圖 **1.** 89-96 年畜牧業搭排數量統計圖

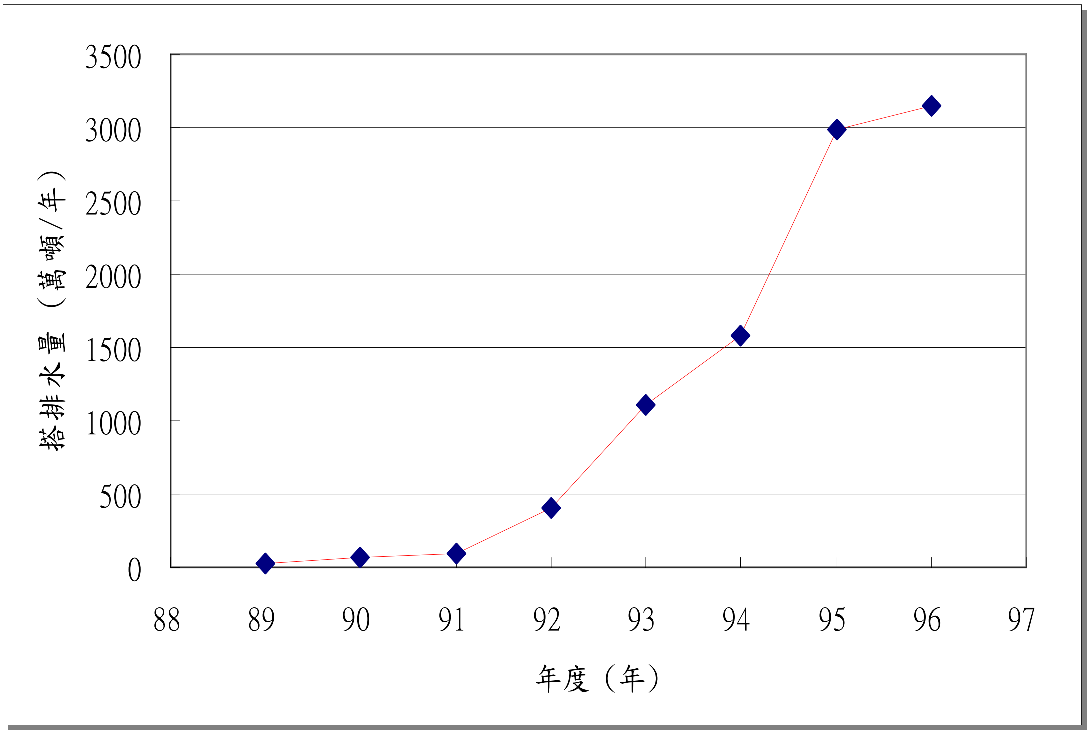

圖 **2.** 89-96 年畜牧業申請搭排廢水量統計表

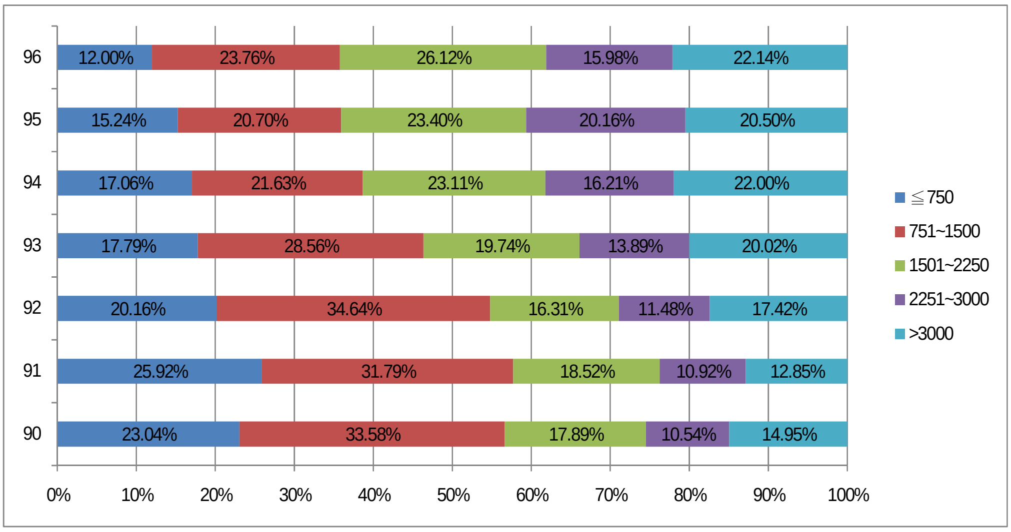

圖 3. 90-96 年全省水利會畜牧排放廢水水質初驗電導度不合格率分析

表 1. 89-96 年遭受畜牧污染渠道水質 EC 統計表　　單位: μS/cm

會別	檢驗數 (點次)	89	90	91	92	93	94	95	96	EC 比較
宜蘭	1211	345	327	327	351	336	330	337	330	-15
北基	251		169	176	238	179	206	197	201	+32
桃園	40	720		603	741	600	631	617	547	-173
苗栗	47	356	408	461	476	531	726	624	642	+286
台中	300	252	281	286	358	353	334	351	358	+106
南投	125	623	440	309	378	557	351	385	348	-275
彰化	2477	848	798	804	780	781	745	831	862	+14
雲林	705			853	760	806	798	734	832	-21
嘉南	656	709	824	1297	1392	1589	811	726	578	-131
高雄	747	1116	1209	1309	1043	1018	1303	1225	1028	-88
屏東	680		658	704	714	778	825	883	917	+259
台東	42	390	380	412	373	403	417	411	415	+25
花蓮	144	363	363	332	373	366	388	399	366	+3
總計	7425									

資料來源:灌溉水質監測管理系統監視點相關畜牧污染測點水質 EC 資料

表 2. 92-96 年度畜牧影響渠道水質複驗項目合格數(率)統計表　　單位:點次

水利會	點數	水溫	pH 值	導電度	懸浮固體	氯化物	硫酸鹽
宜蘭	4	4.0	4.0	4.0	4.0	4.0	4.0
桃園	3	3.0	2.0	0.0	3.0	3.0	3.0
彰化	104	104.0	102.0	15.0	80.0	92.0	97.0
雲林	52	40.0	52.0	27.0	46.0	39.0	48.0
嘉南	183	183.0	182.0	111.0	171.0	156.0	162.0
高雄	41	21.0	40.0	17.0	38.0	33.0	41.0
屏東	18	15.0	18.0	11.0	17.0	18.0	17.0
合計	405	370.0	400.0	185.0	359.0	345.0	372.0
合格率		91.4	98.8	45.7	88.6	85.2	91.9
不合格率		8.6	1.2	54.3	11.4	14.8	8.1

續表 2. 92~96 年度畜牧影響渠道水質複驗項目合格數(率)統計表　　單位:點次

水利會	點數	總氮	銅	鋅	鈉吸著率	殘餘碳酸鈉
宜蘭	4	3.0	4.0	4.0	4.0	3.0
桃園	3	0.0	0.0	3.0	3.0	2.0
彰化	104	73.0	101.0	101.0	93.0	84.0
雲林	52	44.0	52.0	52.0	39.0	35.0
嘉南	183	72.0	175.0	175.0	169.0	165.0
高雄	41	14.0	41.0	40.0	37.0	37.0
屏東	18	9.0	18.0	18.0	18.0	16.0
合計	405	215.0	391.0	393.0	363.0	342.0
合格率		53.1	96.5	97.0	89.6	84.4
不合格率		46.9	3.5	3.0	10.4	15.6

資料來源：灌溉水質監測管理系統監視點相關畜牧污染測點水質複驗數據經本文統計資料。

Evaluation of Nitrate Contamination Potential in the Groundwater Catchment Area of Nan-Tou

Chen-Wuing Liu[1,2]

Abstract

Public concern over groundwater quality has grown significantly in recent years. Agriculture is considered as a major contaminant source of groundwater. Increasing evidence shows that the nitrate contamination exceeds the maximum contaminant level of groundwater in the upper Choushui river alluvial fan of Nantou. The objectives of the study were to apply factor analysis to identify factors that govern the chemistry of groundwater, and to determine the source of nitrate pollutant on its pathway using the groundwater quality data and land use type in this area. These results were integrated using DRASTIC to delineate the groundwater water vulnerability area and formulate the strategy for sustainable use of groundwater resources. Eight hydrochemical parameters including pH, EC, Hardness, NH_4^+, NO_3^-, Fe, Mn and S concentrations were considered to evaluate the characteristics of the groundwater by factor analysis. Three main hydrogeochemical factors were determined. Factor 1 has strong positive loadings on Fe, Mn and NH_4^+ which is a common type of the groundwater hydrochemical environment of Taiwan. Factor 2 has strong loadings on Hardness and EC representing a hardness water property. Factor 3 has strong positive loadings on nitrate and pH which is polluted probably from agriculture, industry and domestic sewage. Redox is the most plausible process determining the groundwater characteristics in this area where the groundwater has oxidized, acidic and high nitrate properties in Nan-Tou city and exhibits reduced, alkaline and low nitrate properties in Dou-Lio. Factor analysis shows the chemistry of groundwater is stable based on the data of 2004-2006. Contamination potential of nitrates is evaluated by DRASTIC. The score exceeds 160 define the scopes of groundwater vulnerable to pollution. The high contamination potential areas of nitrate are Pu-Li, the west of Tsau-Tuen, the southwest of Ming-Jian, the east of Ju-Bei and Nan-Tou city. These areas were agricultural active and densely populated. The nitrate pollution has no immediate threat to the downstream area of Choushui river alluvial fan. However, the present groundwater quality is not suitable for potable water use. In order to achieve sustainable use of groundwater resource, we should plan to remediate the nitrate contamination, construct sewer systems, reduce the use of agrichemicals and protect the areas of high contamination potential.

Keywords: Nan-Tou**,** Factor analysis**,** Contamination potential of nitrate, Groundwater, DRASTIC

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集水區上游南投地區地下水硝酸鹽氮污染潛勢評估

劉振宇 [1,2]

摘要

集水區上游地下水水質之良窳，直接影響中下游地下水使用之安全，近年來濁水溪沖積扇上游(南投區域)地下水因農業盛行導致硝酸鹽氮污染趨於嚴重，逐漸影響中、下地下水之利用，本研究以多變量統計方法之因子分析，評估本區地下水水質結構、特徵及其在空間及時間上之分佈與變化，並藉由現地水質調查及土地使用情況，研判潛在硝酸鹽氮污染源及可能之地下水污染途徑。然後結合 DRASTIC 模式，評估與劃定本地區地下水易受污染區，最後提出永續經營本區地下水使用方案。八項水質包括酸酸値、總硬度、電導度、氨氮、硝酸鹽氮、鐵、錳及硫化物爲因子分析之數據，本區域 2004~2006 年地下水水質因子分析研究果穩定。鐵錳及硬度因子爲台灣地下水水質特徵之一，皆爲地層中溶出，而硝酸鹽氮則爲地表人爲活動而導致；南投市之地下水水質呈氧化、酸性及高硝酸鹽氮之水質特徵，雲林斗六之地下水水質呈還原、中鹼性及低硝酸鹽氮之特徵；DRASTIC 模式評估本區之硝酸鹽氮地下水之污染潛勢結果，污染潛勢得分在 160 分以上之區域劃定爲易受污染區，其中呈現高硝酸鹽氮汙染潛勢包括埔里鎮、草屯鎮西半部、名間鄉西南側、竹山鎮北側及南投市內零星區域，皆爲南投縣之人口集中或農業盛行之區域。南投縣境內之硝酸鹽氮污染已超過 10 年以上且爲多重污染源，短期內雖未影響濁水溪沖積扇及台中盆地中游地下水區，但本區之地下水水質條件已不適民生用水之需。鄉鎮市應積極推動地下水污染防治，人口密集之鄉鎮市應加速興建衛生下水道，農業發達之鄉鎮應管理並輔導農戶減少使用農藥及化肥，以期達成地下水資源保育與永續利用之目標。

關鍵詞：南投、因子分析、硝酸鹽氮污染、地下水、DRASTIC

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前言

集水區上游水質之良窳，對中下游地下水質影響重大，濁水溪沖積扇及台中盆地之上游為南投地區 (以下簡稱本地區)，境內地勢除西側南投市、草屯鎮、名間鄉等為平原地形外，其餘鄉鎮多為臺地或山地地形，地形高差變化極劇，且本地區因位於地面水水源之上游區域，地表水資源供應充足，地下水資源於本地區不似彰化、雲林縣及台中盆地等西南縣市獲主管機關重視，因而經濟部水利署建置之地下水觀測站網多以平原地下水區為主，僅部份觀測井設置於濁水溪沖積扇上游。環保署之觀測井多屬場置性污染之水質監測功能，南投縣政府自民國 91 年起逐步設置特定污染源地下水監測井達 22 口，由監測井之水質結果均顯示地下水氮污染情形逐年惡化。台灣省自來水公司原自南投名間抽取地下水作為自來水水源，也於該年改抽取水里溪之河水替代，顯示硝酸鹽氮污染已嚴重影響民生用水水源，必須盡速評估本地區地下水氮之污染潛勢及因應方案。本研究目的為調查、分析南投地區地下水水質變化、結構及特徵，採用 DRASTIC (US EPA, 1985) 評估本區污染潛勢，及提出有效之管理策略。

研究方法

研究區域

本區位居本島中央，台灣地理中心碑設於本縣埔里鎮，是全省唯一不濱海之縣份，面積遼闊，南北長約九十五公里，東西寬約七十二公里，境內地勢除西側南投市、草屯鎮、名間鄉等為平原地形外，其餘鄉鎮多為臺地或山地地形。本區內有台中地區與濁水溪沖積扇兩個主要地下水區，其地下水來源主要來自濁水溪沖積扇上游、台中盆地及埔里盆地等三區域 (圖 1)；楊萬全等 (1976) 曾推估埔里盆地之地下水流入量約為每年 124×10^6 噸，約合 3.93 CMS，因此地下水蘊藏量相當豐沛，但經濟部水利署所建置之地下水觀測網，於本地區境內除竹山鎮外，並無針對本地區地下水資源特性 (水量及水質) 進行例行之觀測，所以對於本地區地下水流場特性及水質變化行為方面，各界所知甚為有限。

1997~2006 年水利署水文水資源資料庫中 13 口水位觀測井分別位於台中盆地及濁水溪沖積扇兩大地下水區，分析南投地區地下水位及水位歷線變化，各井每年平均水位之標準偏差值差異性較大，表示本區域之觀測井水位變動較為明顯，且地下水水位特性為易受到豐枯水年而產生變動，則表示該地區地下水層滲透性佳、補注特性良好，為主要之地下水補注區；以埔里盆地地下水位歷線結果而言，水位歷線在夏季豐水期水位抬升，在枯水期水位下降，

乾溼變化最爲明顯，因此將豐水期間多餘之地面水引入沖積扇上游補注區內適宜之含水層，增加其補注量，作爲枯水期間抽取，以供各標的不足用水之使用，可達到地面水及地下水資源之聯合運用。

爲瞭解本區地下水流向，本研究蒐整經濟部水利署及南投縣政府之南投地區地下水水位資料，繪製 2003 年本區地下水等水位線之空間分佈 (圖 2)，南投地區地下水源起於中央山脈流至濁水溪沖積扇上游及台中盆地，流向大致以東南流至西此外，本區域之埔里盆地因地形之條件，地下水型態爲小區域型地下水區，且爲蘊含較豐富地下水之最佳儲水位置，此外名間地區也呈現類似於小區域型地下水區之型態。

本區地下水水質硝酸鹽氮、氨氮及硬度之濃度有普遍偏高情況，爲南投縣內首要追蹤之標的。本研究彙集濁水溪沖積扇上游地下水 (雲林、彰化、南投及台中) 2004~2006 年之 81 口觀測井水質資料觀測井分佈如圖 3，作爲南投縣地下水水質時空分佈特性之探討。

2004~2006 年之地下水硝酸鹽氮濃度分佈 (圖 4)，三年之平均濃度 (3.3、5.8 及 5.5 mg/l) 呈現升高趨勢，且高硝酸鹽氮污染皆集中於南投市區及名間鄉，淺層的硝酸鹽氮濃度高於深層，淺層地下水受到硝酸鹽氮污染較爲明顯，其穿透之深度範圍約在 52~125.5 m，並有超過管制限值的疑慮。2004~2006 年南投市之數口觀測井曾出現高濃度之硝酸鹽氮污染，由於附近產業結構以農業用地爲主，種植水稻、鳳梨、茶葉及生薑等作物，並盛產茶葉聞名，過量施用肥料可能導致高硝酸鹽氮污染，然而目前硝酸鹽氮濃度分佈尚未見明顯污染擴大，但高污染源接近於濁水溪沖積扇頂之區域，爲透水性良好之地下水補注區，未來硝酸鹽氮汙染可能有擴大之虞。本區域氨氮濃度分佈 (圖 5)，三年之平均濃度 (0.3、0.2 及 0.19 mg/l)，呈現略減的趨勢，氨氮高污染中心皆分佈在彰化、雲林一帶。本區域硬度濃度分佈 (圖 6)，三年之平均濃度 (257.9、248.4 及 232.1 mg/l)，但皆超過飲用水水質標準 (150 mg/l)，2005 及 2006 年於彰化淺層觀測井螺陽國小 (50)、柑園 1 (26) 之硬度分別爲 580.3 及 605.1 mg/l，顯示淺層硬度濃度偏高的趨勢，隨觀測井深度增加而降低，與硝酸鹽氮之垂向分佈相似，推測地下水硬度之來源爲地表或淺層地層。

自來水井水質檢測

主要選取南投縣境內台灣省自來水公司之水井，全區共 19 口水井，其中 11 口抽水/加壓井爲南崗工業區附近，另 10 口則靠近南投市區，以供居民作爲民生用水 (民國 91 年 9 月以前)，由於台灣省自來水公司之觀測井無法測得地下水位，因此現場僅進行量測水溫、pH 值及導電度至少五次以上，直到最後連續三次各項參數符合採樣方法所規定之穩定標準，取得

新鮮之水樣。所採水樣依檢驗項目分別盛裝於無菌袋、100 公升採樣瓶 (PE 瓶)，隨即將水樣置於冰箱中，並於當天將水樣利用全程低溫運至檢驗室分析，分析項目爲 pH、氨氮、硝酸鹽氮、鐵、錳、硬度、鈣及鎂。分析方法採用環保署公告之檢驗方法 (行政院環保署環境檢驗所 http://www.niea.gov.tw/)。

因子分析

本研究利用因子分析法 (Factor analysis) 進行水質資料分析，其目的爲尋找原來變數的線性組合使其變異數最大，進而將多個具相關之變數簡化爲少數幾個無相關之變數，若變數間皆不相關則無需利用此方法再簡化，屬一探索性資料分析方法。因子分析法與回歸分析最大之差異爲此法中每個變數皆視爲同等，而迴歸分析爲將變數分成自變數與應變數兩類，而因子分析的優點是能降低維度，即減少變數個數，只考慮幾個主因子而不計權重較小之因子，但缺點則是會損失少部份原始資料代表的訊息。而因子分析法之分析步驟簡述如下 (羅積玉，1994；陳順宇，1998)：

1. 原始資料之標準化

   由於各個變數之單位不同，故先將原始變量數 (15 項檢測項目) 與觀測樣本數 (100 口觀測井) 所構成的原始數據矩陣予以標準化轉換，標準化轉換後之數據矩陣，其平均值爲 0，變異數爲 1。

2. 相關矩陣之主成分分析

   由上述過程所產生之相關矩陣求解其特徵值及其相對應的特徵向量，再由單位特徵向量矩陣與上述之標準化數據矩陣，可得一組由原始變量所重新組合且彼此互不相關的新變量矩陣，然後其進行標準化處理。

3. 確定主因子

   由於因子分析法中各個變數皆爲同等，故由上述兩步驟可得數個新因子，且其數量與原始變量數相同，此時必須對這些因子進行刪減，且保留重要主因子以簡化問題，而主因子選取的方式一般有四種，包括陡坡測試 (scree test)、保留特徵值大於 1 之因子、因子特徵值之解釋能力小於 5 %予以刪除，或採用前面數個變異量總和達到全部變異量總和的 80-95 %之因子。

4. 因子軸旋轉

   爲使各個主因子與變數間關係更爲清楚，須將上述過程中產生之因子負荷矩陣進行旋轉，使因子負荷趨向 0 或 1 兩極化靠近，達到結構簡化與主因子意義清楚之目的。而

旋轉方式眾多，本研究使用最常見之變異最大旋轉法 (varimax rotation)，使因子負荷矩陣中，各因子負荷的總變異量達到最大。

5. 因子得分計算

最後得到之主因子為原始變數之線性組合，故利用因子得分計算可由因子的角度解釋原始變量，並進行歸納。將轉置後的因子負荷矩陣、相關矩陣之逆矩陣與標準化後之數據矩陣三者相乘即為因子得分。

由於南投地區地下水觀測井水質數據有限，因此再將雲林、彰化、台中等地地下水觀測水質數據，一併加入探討。水質項目包括酸鹼值 (pH)、總硬度 (Hardness)、導電度 (EC)、氨氮 (NH_4^+-N)、硝酸鹽氮 (NO_3^--N)、鐵 (Fe)、錳 (Mn)及硫化物 (S) 共計 8 項，作為因子分析 (Factor Analysis) 項目，並以選近三年 2004~2006 年進行因子分析，探討本地區之水質特徵及污染指標並繪製主要因子之空間分佈，因子分析軟體採用 SPSS。

DRASTIC

DRASTIC 為評估一地下水區之易受污染之空間分布模式，易受污染區經 DRASTIC 評估劃定後可作為後續地下水污染防治工作之依據，DRASTIC 考慮七項地文與水文地質參數，以評估一地區受污染潛勢之程度。同時，由於此七項水文地質參數均可以地理資訊系統 (ArcGIS 8.3) 之地圖形式表現，將一地區之污染潛勢程度顯現於圖面上。DRASTIC 系統除對一般性污染物提出考慮水文地質參數之一般性權重外，尚針對農業行為提出農業權重。DRASTIC 污染潛勢評估系統實際應用在南投地區時，所使用的七項參數之給定分數條件皆為同一標準，所以從圖上所獲得之指標值僅提供地下水污染潛勢間相對之程度大小關係，並不能指出污染潛勢之絕對大小值，且參數之給定範圍為 1~10，因此為一指標性之評估模式，此為 DRASTIC 模式之限制。

DRASTIC 七項水文地質參數為:

D: 地下水位深度 (Depth to water)

R: 淨補注量 (Net Recharge)

A: 含水層介質 (Aquifer media)

S: 土壤介質 (Soil media)

T: 地形坡度 (Topography)

I: 未飽和層影響 (Impact of vadose zone)

C: 含水層水力傳導特性 (Conductivity of the aquifer)

DRASTIC 系統將考慮之七個水文地質參數分別轉換成 1~10 分之分數等級，前述七項水文地質參數皆可用地圖形式來表現。

過去交通大學土木工程研究所 (1999) 及洪瑛鈞 (2002) 曾在台灣濁水溪沖積扇與金門地區進行 DRASTIC 污染潛勢評估，本研究以 DRASTIC 模式劃定南投縣地下水易污染區，以探討地表農業、工業及生活污水之硝酸鹽氮污染可能入侵含水層之區域，並將劃定污染區域與目前地下水硝酸鹽氮污染範圍進行比較；探討硝酸鹽氮之污染潛勢，並建議本區地下水污染防治之方案，已確保本區地下水之永續利用。

結果與討論

地下水水質特性分析

自來水水質分析

台灣省自來水公司共 19 口採樣井檢測 pH、氨氮、硝酸鹽氮、鐵、錳、硬度、鈣及鎂。水質檢驗結果與飲用水水源水質標準、灌溉水水質標準及地下水水質監測基準比較結果如表 1 所示。依據地下水監測基準為 5 mg/L 及地下水管制標準、飲用水水源標準為 10 mg/L，此次分析 19 樣本共有 9 項水樣超過飲用水水源水質標準，且共有 18 項水樣超過地下水管制標準，僅有一處之硝酸鹽氮濃度低於管制標準。水質分析結果顯示在本區地下水已不適合作為飲用水質之來源。

因子分析

中央地質調查所之「台灣地區地下水觀測網整體計畫」第一期 (81~87 年度) 濁水溪沖積扇共完成 88 站水文地質鑽探，鑽探總深度為 17,051.5 M，將濁水溪沖積扇劃分為四個含水層，以四個阻水層區分。南投區域 (濁水溪上游等地) 含水層一深度為 19~103 M 之間，最深之深度為 103 M 平均厚度約 42 M，含水層二深度介於 35~217 M 之間，含水層二分布為最廣，含水層三深度為 140~275 M，含水層四深度大於 275 M。因此本研究在水質因子分析中將分為兩種方式進行，一為不分深淺層將所有觀測井進行分析，其二依據定義淺層深度為 9~100 M，深層深度為 100~260 M，將代表淺、深層地下水水質進行分層水質特徵之探討，分別繪製各因子得分分布圖作為比較。

2004 年 (該年水質數據較為完整) 濁水溪沖積扇及台中盆地上游 (包括南投、雲林及彰化等地區) 地下水之因子分析結果顯示，因子 1 解釋總變異量達 35.9 %(表 2)，其中與鐵 (Fe)

及硬度 (Hardness) 呈高度相關性，而導電度 (EC) 及錳 (Mn) 呈中度相關 (表 3)。硬度 (Hardness) 乃由多種多價陽離子所引起，主要爲鈣與鎂，其他如鋁、鐵、錳、鋅等亦會增加硬度，且地下水因與地層中之礦物反應，導致水中硬度升高；鐵及錳爲岩石及土壤的組成成分之一，且由於地下水與地層礦物之交互作用，致使地下水之鐵與錳含量會較高 (張介翰，2006)；此外，當地下水流入地層後，開始發生還原反應，漸漸將溶氧消耗殆盡，因此，導致地下水硝酸鹽氮濃度漸減及鐵錳濃度逐漸升高，而南投地區地下水多屬此弱還原環境，所以有利於部分硝酸鹽氮、二價鐵及二價錳存在 (彭宗仁等，2001)。故因子 1 可命名爲「硬度及鐵錳因子」其空間分佈圖 7 所示；水質之軟硬與否，於飲用水中之主要影響爲味覺口感，對於公眾健康影響不大。由 34 口觀測井 (9~200 M) 之硬度及鐵錳因子得分結果，溪州 1 (34) 爲地下水水質最硬之地區如圖 7 所示，與淺井 (9~100 M) 之分析結果相同如圖 7(b) 所示，其溪州 1 (34) 之總硬度濃度爲 1060 mg/l (以 $CaCO_3$ 爲基準)，但由深井 (100~200 M) 之硬度及鐵錳因子潛勢如圖 7(c) 所示，六合 (29) 爲地下水水質最硬之地區，其總硬度濃度爲 560 mg/l (以 $CaCO_3$ 爲基準)，因此，濁水溪沖積扇上游 (南投地區) 地下水水質硬度因子空間分佈，可依據含水層深度不同，而產生水質硬軟程度也不同，含水層深度爲 20~50 M 水質硬度程度較爲明顯，故可推估淺層地下水易受到地面垂直入滲溶出造成硬度升高。

因子 2 解釋總變異量達 18.19 %（表 2），其中與 pH 及 NO_3-N 呈高度相關性，因氨氮經氧化作用形成硝酸鹽氮的過程中，會使 pH 成酸性反應，以致因子得分圖兩者呈負相關如表 3 所示；且南投地區因其農業活動頻繁，茶園旱作密集分布，大量使用人工肥料、動物排泄物，導致非點源氮化合物 (氨氮、硝酸鹽氮及亞硝酸鹽氮) 入滲污染地下水。因子 2 可命名爲「硝酸鹽氮污染因子」 如圖 8 所示，南投地區地下水硝酸鹽氮濃度介於 0.03 至 18 mg/l 之間，由觀測井不分層 (9~200 M) 及淺層觀測井 (9~100 M) 之硝酸鹽氮因子得分結果，新光 1 (6) 爲地下水水質硝酸鹽氮因子得分較高之區域如圖 8(a)(b)所示，本區域之硝酸鹽氮濃度爲 17.75 mg/l 及 pH 爲 5.12，顯示高硝酸鹽氮及酸性之水質特徵。深層觀測井 (100~200 M) 較高之硝酸鹽氮的地區則爲新光 2 (7)如圖 8(c) 所示，以此深度之含水層呈中性 (6.5) 及呈低硝酸鹽氮 (0.03~1.36 mg/l) 污染，較淺層地下水硝酸鹽氮汙染較不明顯，故可推估淺層地下水易受到地面垂直入滲溶出造成硝酸鹽氮污染升高之趨勢。高硝酸鹽氮因子得分區域皆位於濁水溪上游南投市一帶，然濁水溪沖積扇頂爲地下水補注區，其溶氧及含氮的化合物在溶氧的氧化下，易轉變成硝酸鹽氮，若以 0.5 mg/l 爲界高濃度硝酸鹽氮分佈在淺層 (9~120 M) 爲最廣；南投市與名間地下水高濃度硝酸鹽氮污染，其污染源應爲地表中受到人類活動所致，硝酸鹽氮汙

染其潛在的趨勢，與該區域的原生地質、水文、水文地質特性等亦關。南投市與名間之地質結構大多屬於沖積土及紅土組成，一般紅土的特性爲多鐵質、多孔隙，呈非拘限含水層，而各含水層水力傳導特性較佳 (K=0.03~1.9410^{-3} m/sec)，由此可知南投市與名間鄉之水文、地質條件應有良好地下水補注特性，因此易造成地表硝酸鹽氮污染物垂直滲入，且因流通性佳進而影響深層之含水層，由於兩區域之土地利用以農業爲主，因而過量施用肥料，可能導致地表入滲造成高硝酸鹽氮污染，初步推測硝酸鹽氮汙染之主要爲農業施肥及飼料污染所致；根據彭宗仁等 (2007) 顯示名間鄉地下水 NO_3^-濃度無明顯的季節變化且變異係數皆不高，因此 NO_3^-污染主要是受到長期農業活動所影響且 NO_3^-污染來源穩定，與本研究結果相符。因子 3 解釋總變異量達 12.76 %（表 3），主要爲硫化物 (S)，其餘皆呈低度相關性，因子 3 可命名爲「硫化物因子」，由於南投地區受硫化物污染的可能性極低，且硫酸鹽濃度皆未超過飲用水水源水質標準（250 mg/L）， 2005 及 2006 年皆未顯現故對水質影響不大。

爲進一步探討時序變化對於南投地區 (包括雲林及彰化等地區) 地下水水質之特性及關聯，分析 2004~2006 年之因子得分分佈，可瞭解三年間此區域不同年份與各分層之水質變化及水中化性的穩定程度。2004~2006 年以各因子與水質項目之負荷關係，取較高負荷之水質項目如表 4 所示，三年之因子分析結果顯示本區域地下水水質特性呈穩定現象，主要影響之水質特徵相似度高，仍以鐵、錳、硬度及硝酸鹽氮因子爲本區域水質特徵。結合 2004~2006 年淺層及深層地下水水質特性及因子分析結果，並瞭解南投地區地下水可產生氧化還原分區；由淺層地下水因子分析得分如圖 9 所示，南投市一帶 (深黑處) 爲高硝酸鹽氮污染潛勢區，此區域之水質硝酸鹽氮濃度高且呈酸性表示地下水水質爲氧化的特性，代表三年間南投市 (名間鄉) 一帶較爲氧化，但深層地下水並未有此現象。本區域地下水除淺層氧化水質特性，深層水質尚有還原的特性，由於還原環境易造成鐵、錳的釋出，深層地下水之高鐵錳濃度皆於雲林斗六 (深黑處) 以圖 10 所示，表示此區域之地下水水質趨向還原、中鹼性、低硝酸鹽氮污染特性。

硝酸鹽氮潛勢分析

DRASTIC 評估之污染潛勢如圖 11 顯示，南投縣境內污染潛勢得分較高之區域 (160 分以上) 包括埔里、草屯鎮西半部、名間鄉西南側及竹山鎮北側，將得分在 160 分以上之區域劃定爲易污染區；將 2006 年於南投地區硝酸鹽氮濃度 (共 27 點) 與 DRASTIC 模式中各因子之分數及總分進行迴歸分析，觀測井之硝酸鹽氮濃度與各項因子之分數與總分並無顯著之相關性，雖然高污染濃度區域和高污染潛勢區域無顯著相關，然而在部分高污染潛勢區域（如南

投市）附近出現高污染濃度區域，而南投地區內地下水質監測井主要集中於南投市區及草屯、名間、竹山等鄉鎮，在以上所述區域，地下水位深度 (D)、地形坡度 (T)等項目均有較高之得分 (7 分以上)；並由 D 值得結果顯示，草屯鎮及埔里鎮爲得分高之區域，其 D 值之權重爲 5，換言之，地下水深度較淺之地點，其污染潛勢經過加權計算後較其他因子具更高之影響；而 T 值得分結果顯示，得分在 9 分以上之行政區也位於埔里鎮、草屯鎮及南投市等地；上述得分較高之行政區，包括南投市、埔里鎮、草屯鎮、竹山鎮、名間鄉，除埔里鎮之外其他四個行政區位於南投縣西部，地勢緩和，而埔里屬於盆地地形，盆地中央之坡度亦屬平坦，與上述 T 值得分較高之區域相符合。而其他高污染潛勢地區中，埔里鎮並未設置水質監測井，無長期記錄當地水質之變化，在埔里應增設數處監測井；至於草屯鎮、竹山鎮、名間鄉 (名間鄉硝酸鹽氮污染擴散至地下水深層自由含水層，如圖 2 顯示該地地下水之匯流區) 等地，硝酸鹽氮濃度雖較南投市爲低，亦設有監測井以觀察水質變化，然而由於其高污染潛勢之特性，應持續進行水質之監測與分析，由圖 2 之地下水流向顯示名間、竹山等地之地下水來自於東側鄉鎮（如鹿谷鄉與集集鎮）。針對本區域之農業興盛區域，根據歐洲共同體於 2002 年 4 月所制訂出特定蔬菜中硝酸鹽最大限量標準 (2,000~4,500 mg NO_3/kg)，反觀國內 2005 年環境永續指數 (ESI) 的資料顯示，使用量平均值爲 152.7 公斤/公頃，由於台灣農民勤於耕種，一年多穫，作物複雜指數高，且氣候高溫多濕，肥料流失快，需要多次施肥，而其他國家大多僅需要施肥一次，因此會高於世界之平均值。未來建議本區應管制地下水補注區之產業活動及土地利用、各項開發計畫，限制地下水補注區之農藥施用肥料用量，協助農戶提出肥料管理計畫 (nutrient management plan)。此外於人口密集之鄉鎮市應提高衛生下水道普及率，以降低生活污水入滲所造成之硝酸鹽氮污染，以確保本區地下水永續利用之目標。

結論

本研究藉由濁水溪上游 (南投縣、彰化、雲林及台中盆地上游) 地下水水資源及水質數據之匯整、分析及模擬評估，建立縣區地下水資源及水質結構特性、劃定易污染區及硝鹽氮污染潛勢分析。本區各水井觀測站之水位歷線呈現多爲平穩狀態，各測站在 9 月以前之水位對降雨補注反應較不明顯，但在 9 月及 10 月以後均呈明顯上升，且水位高低依地形高程而變。由地下水位歷線顯示，地下水由東南及東北朝西流動，且於埔里及名間呈區域型地下水區。本區地下水水質以硝酸鹽氮污染較爲嚴重，19 口自來水公司供水井中 9 口水井之硝酸鹽濃度

超過飲用水水源水質標準 10 mg/l。由因子分析結果顯示本區之水質特徵爲鐵錳因子、硬度因子及硝酸鹽氮因子，其中鐵錳及硬度偏高主要源自地層礦物之溶出，而硝酸鹽氮之污染則屬農業活動及生活廢水入滲汙染地下水所致。而依據 DRASTIC 模式進行污染潛勢評估結果，南投縣境內之污染潛勢得分由 0-240 分之間，將得分 160 分以上之區域劃定爲易污染區，其中主要之行政區包括埔里鎮、草屯鎮西半部、名間鄉西南側、竹山鎮北側及南投市內零星區域，上述地區由於地勢較爲平坦，且地下水深度較淺，因此其污染潛勢較南投縣內其他地區爲高，同時南投縣之人口亦多集中於上述區域。

由目前研究結果顯示南投縣境內近 50 %之監測井其硝酸鹽氮濃度高於飲用水水源水質標準 (10 mg/l)，由於南投縣內之地下水監測井大多位於西半部各鄉鎮如草屯鎮、南投市、名間鄉、竹山鎮等地，以地下水保育管理之角度而言，監測井之設置未能涵蓋全區。爲確保南投縣之地下水資源供應無虞，可參考各國先進國家之地下水硝酸鹽污染防治工作與改善方案管制並減少化肥使用量，同時積極興建衛生下水道，減少生活污水進入地下水層，以達到南投縣地下水資源永續經營之目標。

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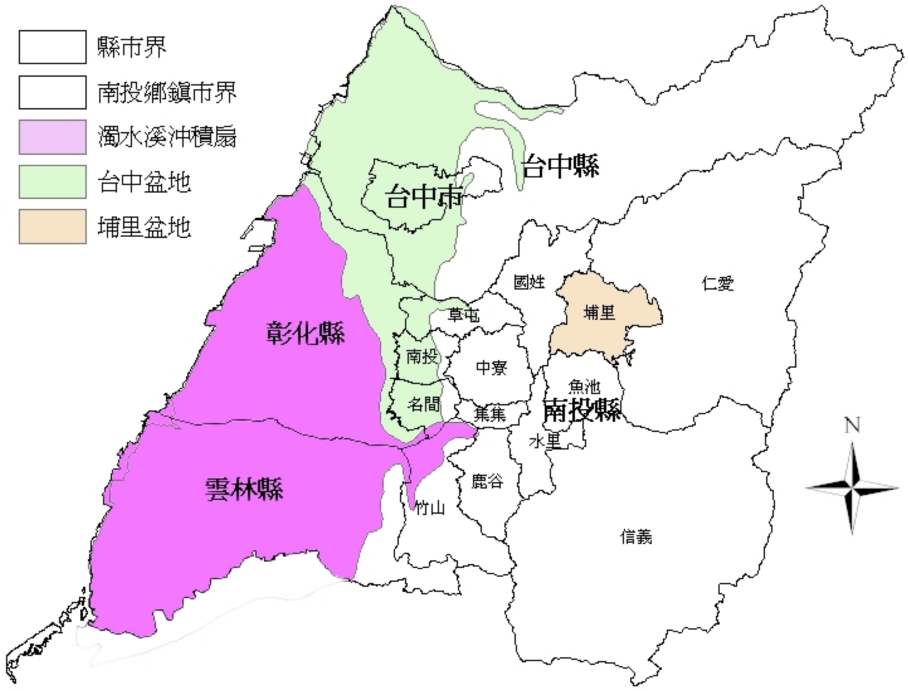


圖 1. 南投縣地理位置及中部地區地下水分區

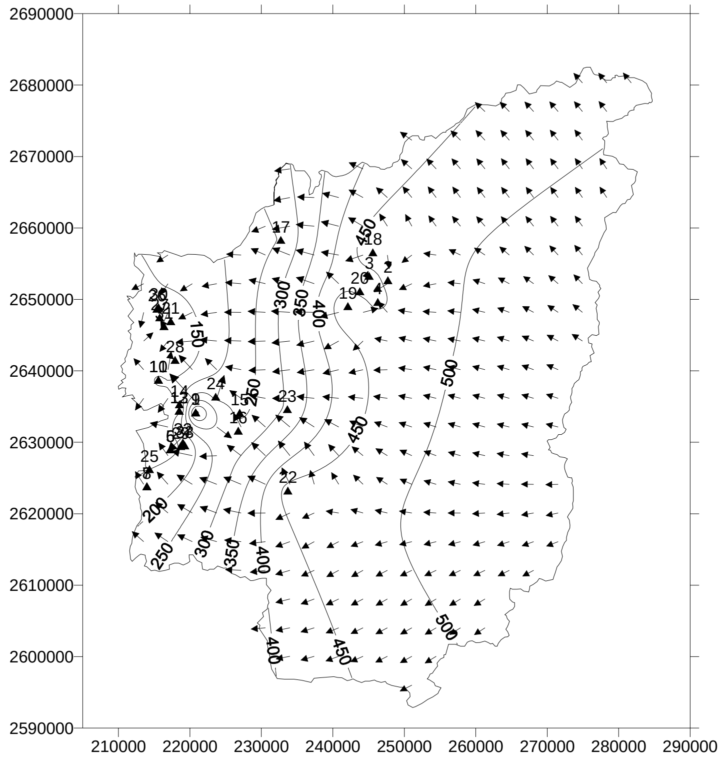


圖 2. 2003 南投縣境內地下水水位及流向(單位 M)

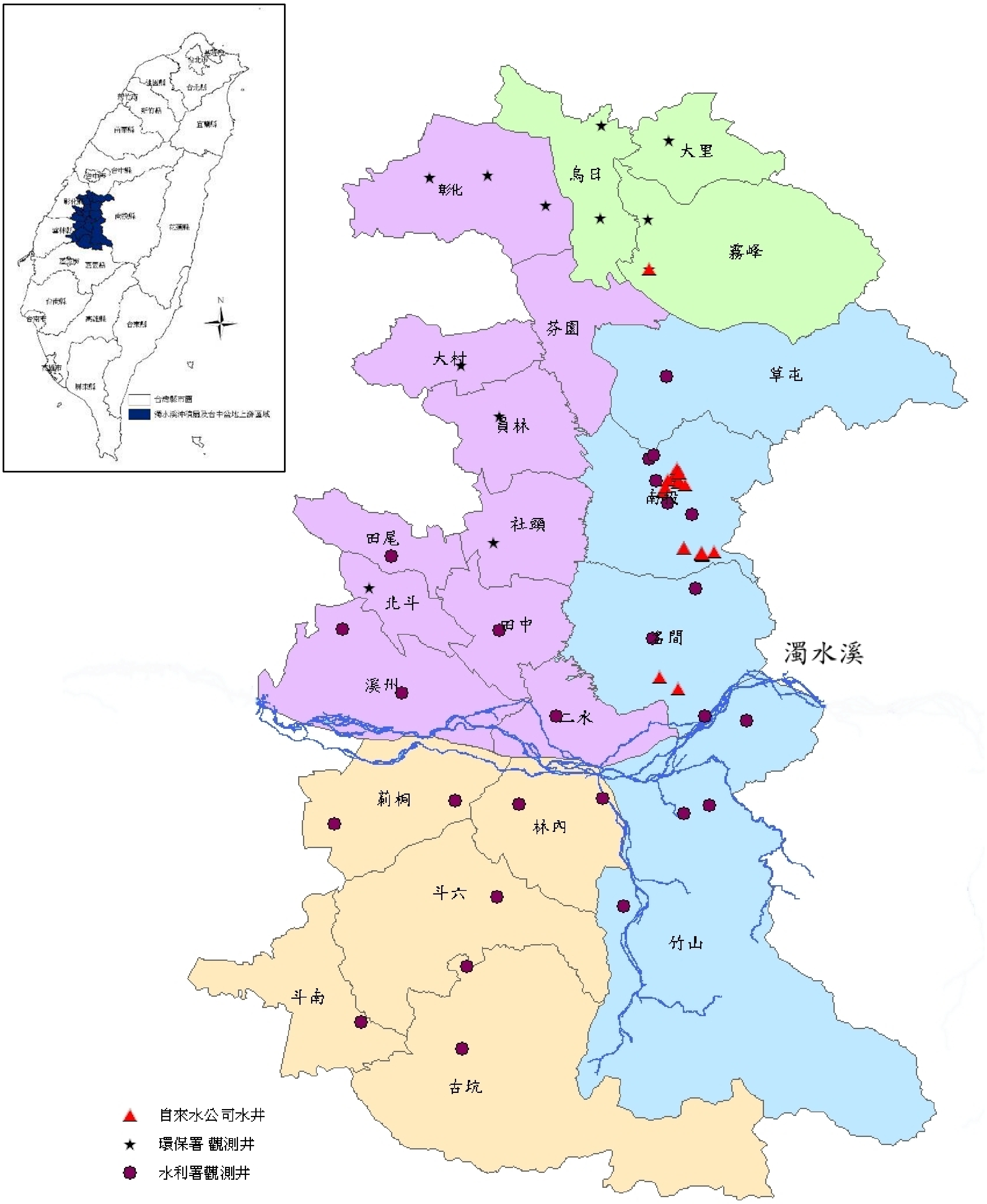


圖 3. 濁水溪沖積扇及台中盆地上游包括南投、彰化、雲林及台中地下水觀測井位置分布

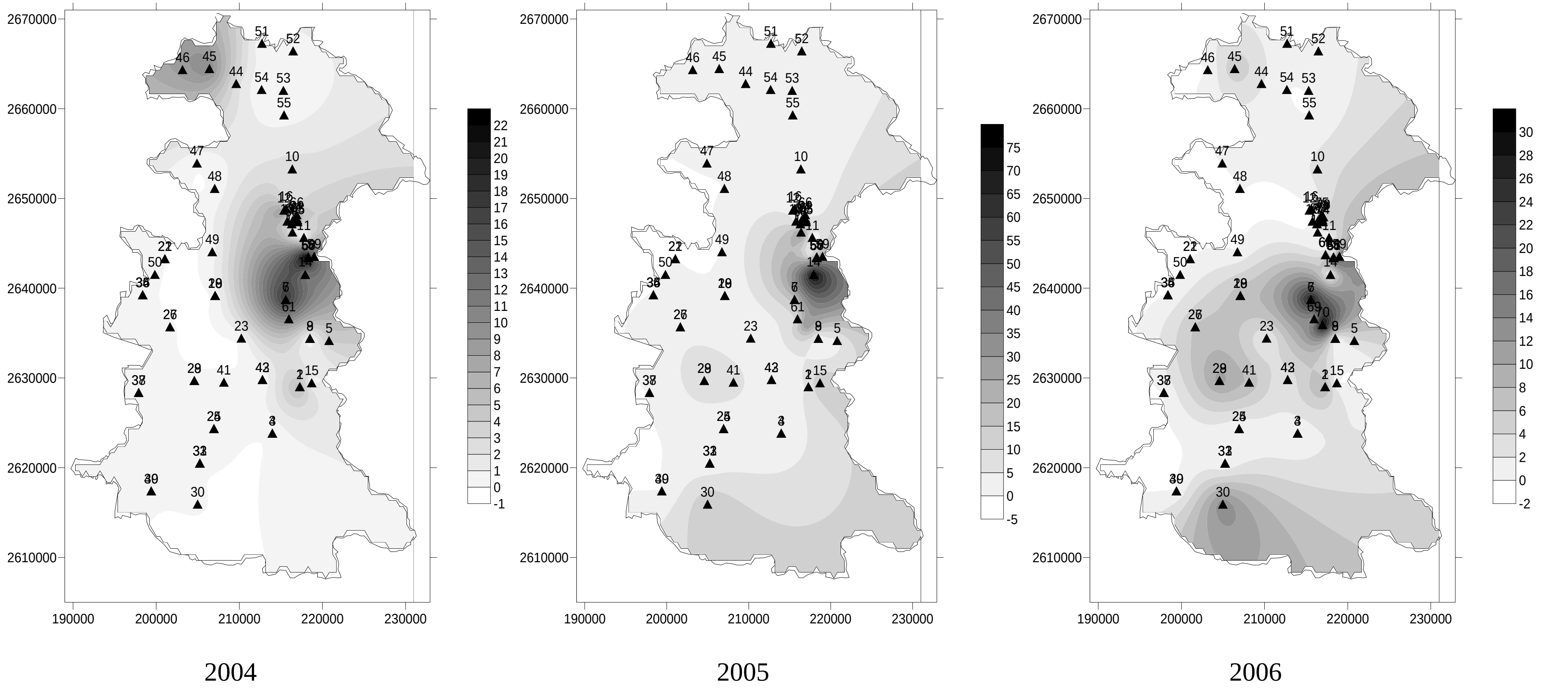
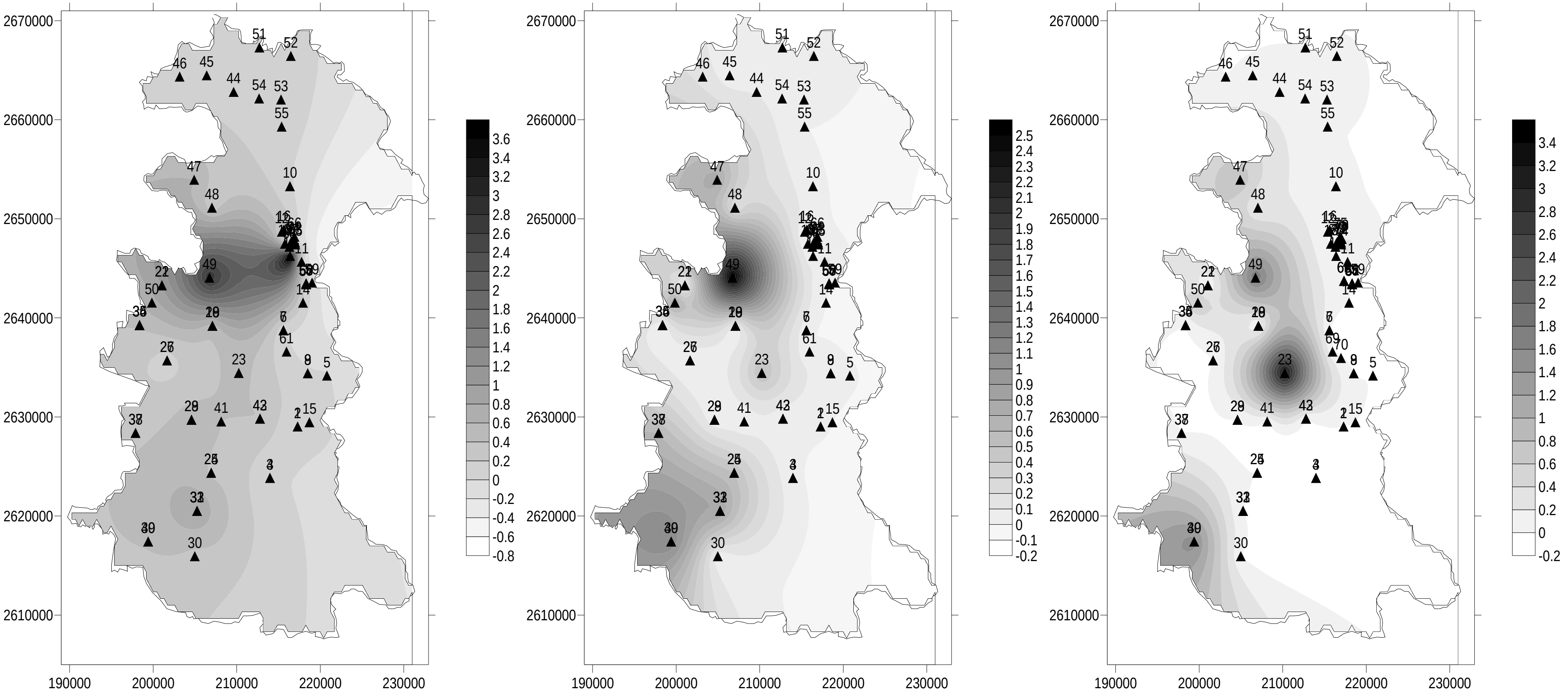


圖 4. 2004-2006 年濁水溪沖積扇及台中盆地上游(包括南投地區)地下水之硝酸鹽氮濃度(ppm)分佈



2004 2005 2006

圖 5. 2004-2006 年濁水溪沖積扇及台中盆地上游(包括南投地區)地下水之氨氮濃度(ppm)分佈

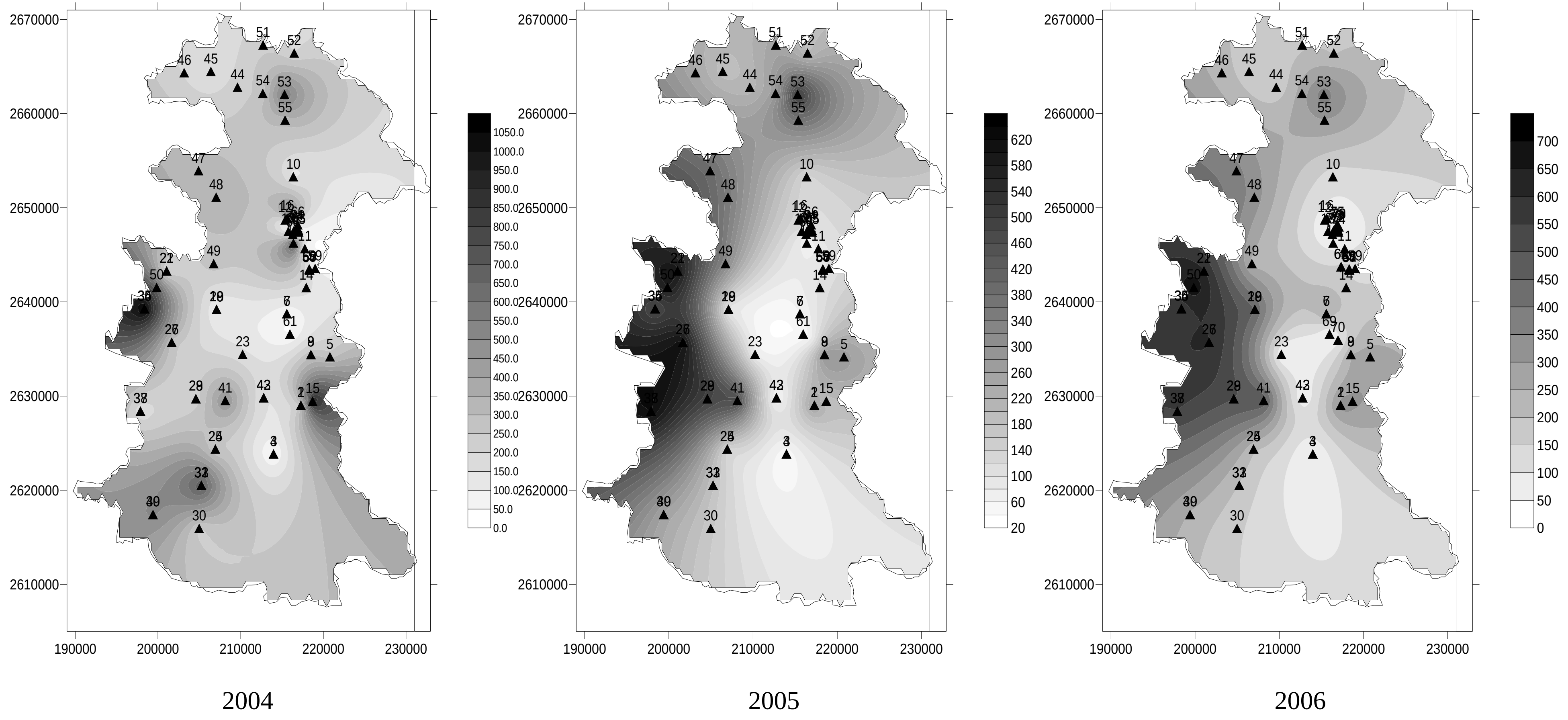
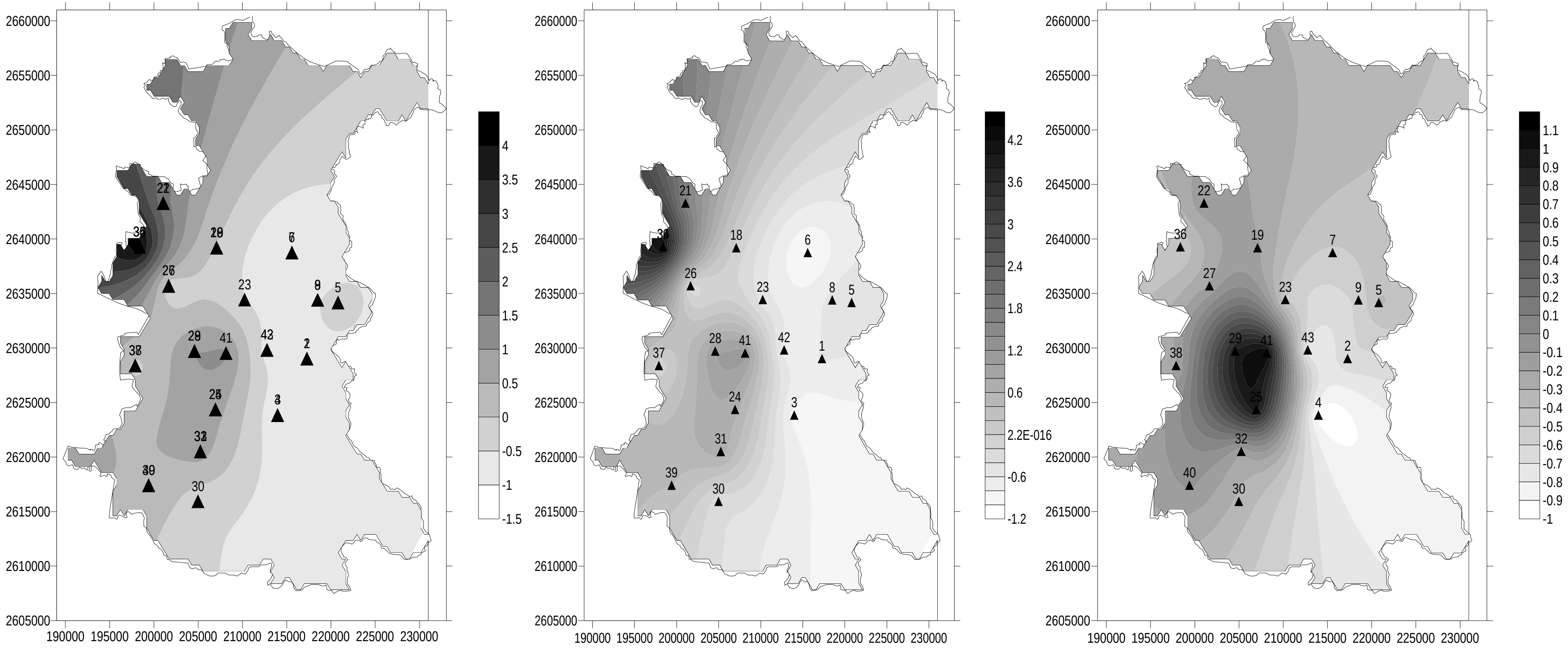
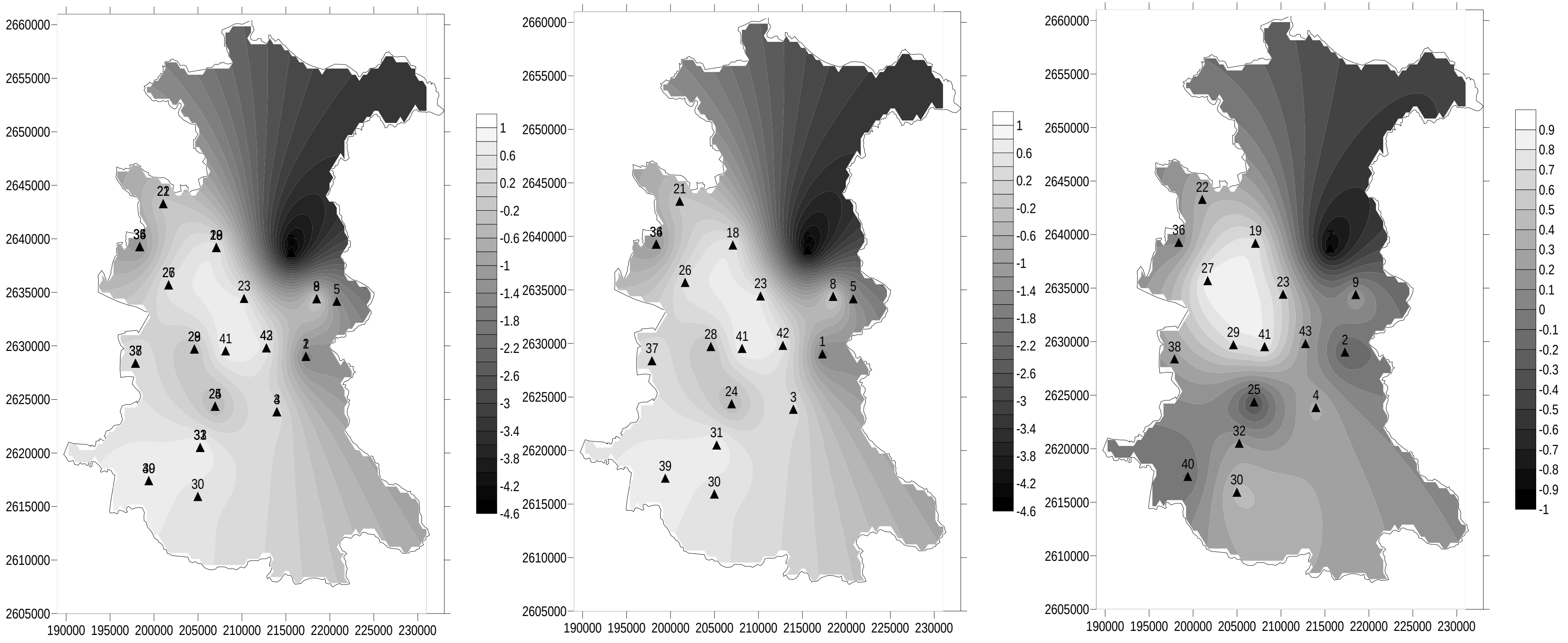


圖 6. 2004-2006 年濁水溪沖積扇及台中盆地上游(包括南投地區)地下水之硬度濃度分佈



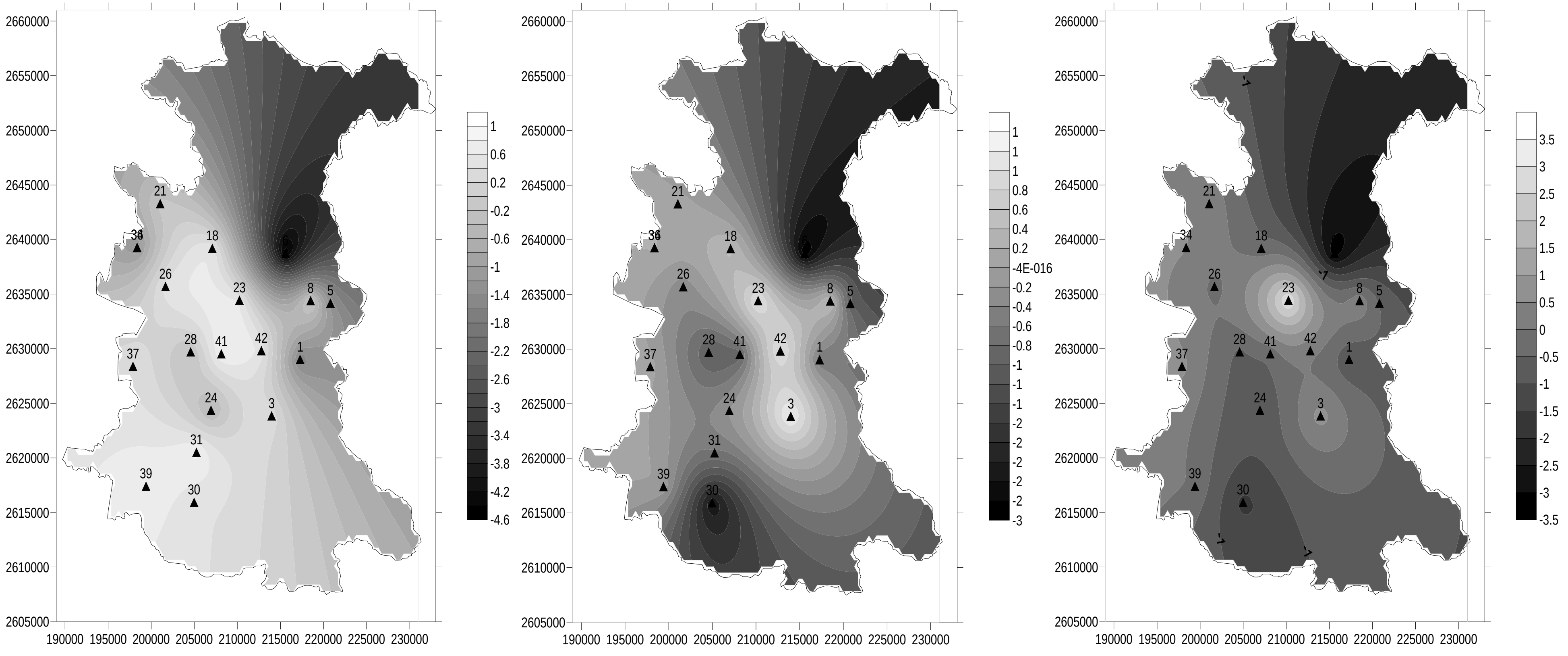
含水層 9-260M　　淺層 9-100M　　深層 100-260M

圖 7. 2004 年之硬度及鐵錳因子得分分布



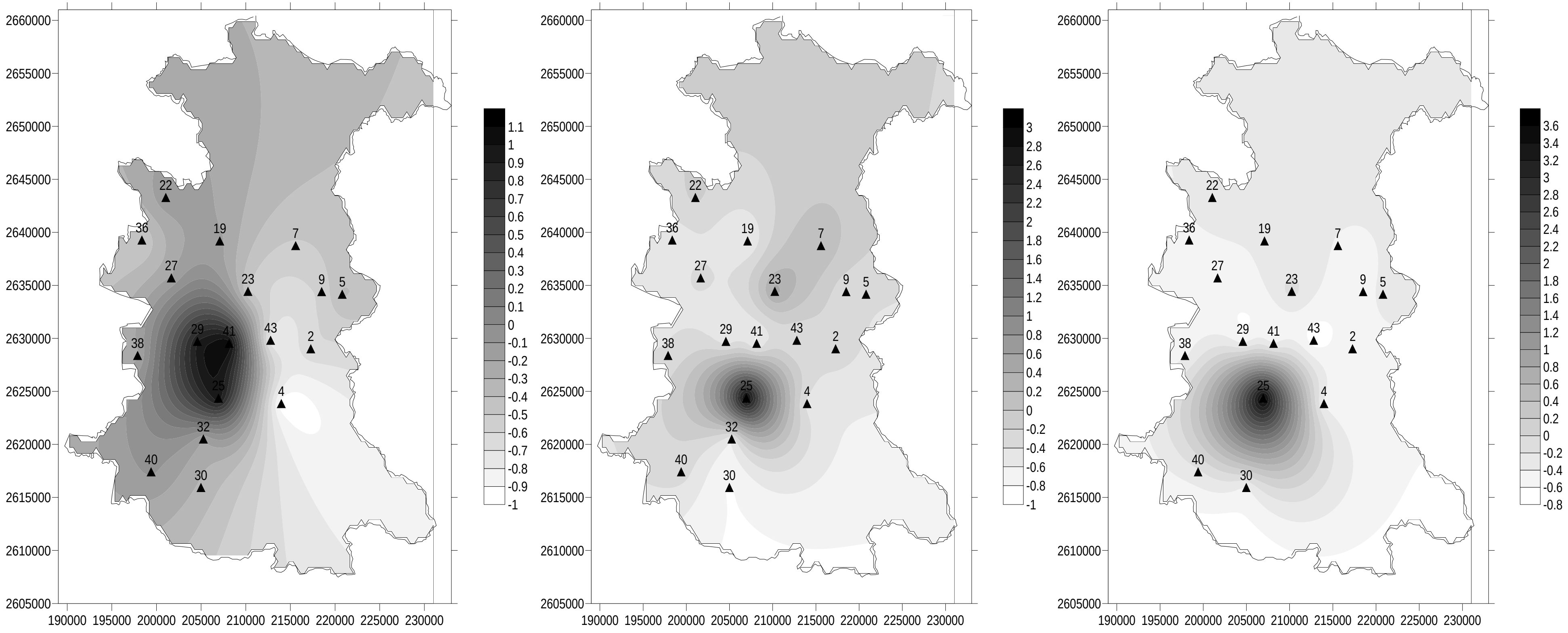
含水層 9-260M　　淺層 9-100M　　深層 100-260M

圖 8. 2004 年之硝酸鹽氮因子得分分布



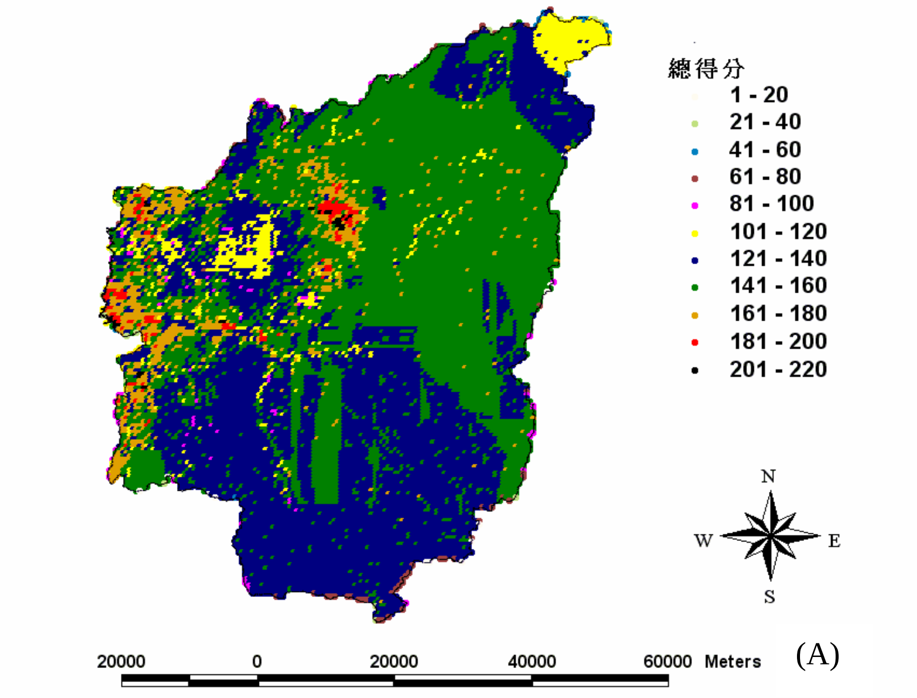
2004 2005 2006

圖 9. 2004-2006 硝酸鹽氮因子得分分布(淺層)



2004 2005 2006

圖 10. 2004-2006 鐵錳因子得分分布(深層)



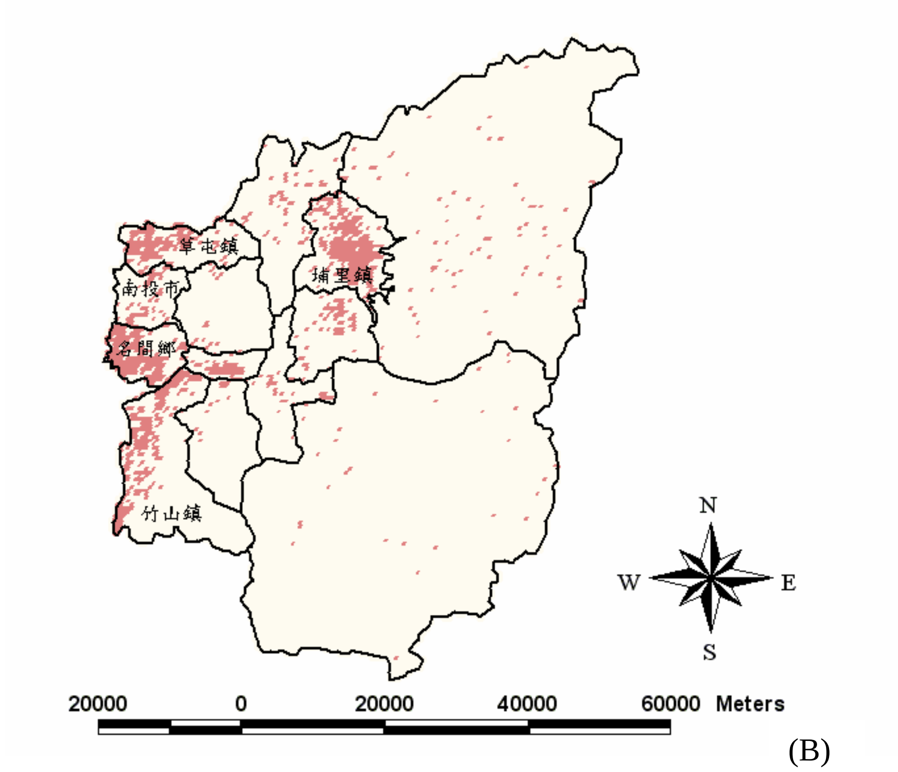


圖 11. 南投地區 DRASTIC 模式之總得分(A)與易污染區之位置(B)

表 1. 自來水水井水質檢驗結果與水質標準比較

檢驗項目	井深	pH 值	硝酸鹽氮	氨氮	鐵	錳	硬度	鈣	鎂
單位	m	---	mg/L	mg/L	mg/L	mg/L	mg/Las $CaCO_3$mg/L	mg/L	mg/L
飲用水水源標準	---	6~8.5	<10	<0.1	<0.3	<0.05	150	-	-
灌溉用水水質標準	---	6~9	-	3 總氮（T-N）	<5	<0.2	-	-	-
地下水監測基準	---	-	<5	<0.05	<0.15	<0.025	150	-	-
嘉和 1	150.0	8.2	13.7	0.05	ND<0.046	ND<0.048	127.0	30.3	12.4
嘉和 2	150.0	8.1	14.6	0.10	ND<0.046	ND<0.048	139.0	33.5	13.5
嘉和 3	150.0	8.0	15.4	0.12	ND<0.046	ND<0.048	133.0	31.8	13.1
嘉和 4	150.0	7.5	19.3	0.08	ND<0.046	ND<0.048	123.0	29.8	11.8
嘉和 5	230.0	7.8	7.3	0.03	ND<0.046	ND<0.048	87.8	22.3	7.8
嘉和 6	170.0	7.5	12.9	0.07	ND<0.046	ND<0.048	70.2	14.6	8.2
嘉和 7	170.0	7.3	10.3	0.10	ND<0.046	ND<0.048	68.5	14.6	7.8
嘉和寬 1	9.7	8.0	14.1	0.50	ND<0.046	ND<0.048	174.0	39.0	18.6
炭寮	200.0	7.3	30.9	0.12	ND<0.046	ND<0.048	91.8	18.5	11.1
崁腳	195.0	7.4	12.6	0.04	ND<0.046	ND<0.048	42.4	8.4	5.2
南崗 1	150.0	7.6	9.1	ND<0.02	ND<0.046	ND<0.048	87.5	17.1	10.9
南崗 2	200.0	8.0	4.6	0.10	ND<0.046	ND<0.048	62.9	15.1	6.1
南崗 3	150.0	7.8	7.2	0.05	ND<0.046	ND<0.048	74.7	15.3	8.9
南崗 4	135.0	7.6	9.0	ND<0.02	ND<0.046	ND<0.048	48.5	8.8	6.4
南崗 11	150.0	7.8	9.6	0.06	ND<0.046	ND<0.048	101.0	19.9	12.5
南崗 15	170.0	7.9	5.7	0.07	ND<0.046	ND<0.048	54.6	11.3	6.4
南崗 16	170.0	7.4	7.6	0.07	ND<0.046	ND<0.048	55.7	10.3	7.3
南崗 17	150.0	7.4	7.1	0.05	ND<0.046	ND<0.048	92.1	18.6	11.1
南崗 18	150.0	7.4	4.9	0.04	ND<0.046	ND<0.048	80.8	17.7	8.9

（灰底）爲超過水質標準之水井水質項目

表 2. 因子分析特徵值大於 1 之因子及其變異數百分比

因子	特徵值	變異數百分比	累積變異數百分比
1	2.87	35.90	35.90
2	1.46	18.19	54.09
3	1.02	12.76	66.86

表 3. 2004 地下水質各因子與水質項目之負荷關係（絕對值大於 0.6 者以粗體表示）

水質項目	因子		
	因子 1	因子 2	因子 3
Fe	**0.906339**	-0.10569	-0.04238
Hardness	**0.808632**	0.056155	0.004856
EC	**0.642308**	0.078933	-0.20457
Mn	**0.635808**	0.014567	0.045523
NH_4-N	0.489944	0.370816	0.160429
NO_3^-N	-0.22202	**-0.89406**	-0.02696
pH	-0.53532	**0.705518**	-0.11984
S	-0.05333	0.001989	**0.967287**

表 4. 2004~2006 年各因子與水質項目之負荷關係

年份	因子分析		水質項目			變異量(%)
2004	因子 1	硬度及鐵錳因子	Fe	Hardness	Mn	35.90%
	因子 2	硝酸鹽氮因子	NO_3^-N	pH		18.19%
	因子 3	硫化物因子	S			12.76%
2005	因子 1	鐵錳因子	Fe	Mn	NH_4-N	34.16%
	因子 2	硬度因子	EC	Hardness		25.57%
	因子 3	硝酸鹽氮因子	pH	NO_3^-N		18.52%
2006	因子 1	鐵錳因子	Fe	Mn		36.74%
	因子 2	硬度因子	EC	Hardness		24.60%
	因子 3	硝酸鹽氮因子	pH	NO_3^-N		16.28%
2004~2006 (平均)	因子 1	鐵錳因子	Fe	Mn	NH_4-N	27.50%
	因子 2	硬度因子	EC	Hardness		24.90%
	因子 3	硝酸鹽氮因子	pH	NO_3^-N		15.75%

Nitrate Pollution of Groundwater in Recharge Area of Alluvial Fans, Taiwan

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Abstract

Alluvial fans are the most important aquifers for its abundant reserve of groundwater resource in Taiwan. Groundwater extractions in alluvial fans of Choshui and Pingtung are over 80 % of total yield in Taiwan. Nitrate levels in Choshui alluvial fan aquifers were higher from 1991 because of agricultural N fertilizer application and untreated sewage leakage. High concentration of nitrate (nitrate-N>10mg/l) in drinking water is known to cause methemoglobinemia (blue baby syndrome) in infants while low concentration of nitrate (nitrate-N 1-10mg/l) is a probable carcinogen. In our monitoring period of 1997-2007, about 10 % of wells in Choshui alluvial fan have elevated nitrate level and a higher value of 32 % were detected in Pingtung alluvial fans because of where with a broader fan head recharge area. There are 7 wells in Choshui with nitrate-N concentrations that once exceed the standard for drinking water of 10 mg/l and 4 wells exceeding the standard in Pingtung. Nitrate concentrations in groundwater show seasonal variation that are lower concentrations in rainy season for the dilution of rainfall and stream water and the nitrate concentrations are higher in dry season with the soil water was evaporated. Denitrification is another important process to control the nitrate concentration under the detection limit in groundwaters of middle-lower fan. Nitrate is one of the world's most common pollutants in groundwater system. During the last 50 years, human activities have dramatically increased inputs of nitrogen to groundwater. We suggest that new comprehensive monitoring program should be implemented to monitor these localized areas on site where nitrate have been detected in groundwater above regulatory guidelines in public-supply wells, domestic wells of local communities and monitoring wells.

Key words: fan head recharge area, Choshui, Pingtung, groundwater, nitrate

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濁水溪與屏東平原地下水之硝酸鹽污染

陳文福 [1,3] 呂學諭 [2]

摘要

沖積扇地層是台灣重要的地下水資源區，其中濁水溪沖積扇及屏東平原沖積扇群，估計地下水占全台抽用量之80 %。飲水中含硝態氮所造成的危害，於高濃度（>10 mg/l）時可能造成孕婦流產及新生兒藍嬰症（缺氧致死）；中濃度（1-10 mg/l）時可能會致癌。監測數據顯示從1991年開始，濁水溪沖積扇扇頂的地下水含有中高濃度的硝態氮，因爲扇頂區礫石層透水性佳且缺乏泥層隔絕，易受農業化肥與生活污水滲入的影響。本研究整理1997－2007年於濁水溪及屏東平原地下水區的監測數據顯示：濁水溪地區約10 %觀測井的硝態氮濃度偏高，多位於扇頂補注區，曾超過環保署飲用水限值10 mg/l的井有：東和、坪頂、社寮、古坑、二水、新光、田中等7口，其中又以新光及古坑井超過限值的次數最多。屏東平原約有32 %的井硝態氮濃度偏高，其井數較多的原因爲屏東平原扇頂礫石層的分布比濁水溪地區更廣。監測十年間有四口井的濃度曾超過限值：萬巒一、建興一、泰山一及新庄一，又以建興一較嚴重。本研究也發現：降雨及河川水的補注，因稀釋而使硝酸鹽濃度降低，但在乾季時、因土壤水份蒸發，造成淺層地下水的硝酸鹽濃度升高。地下水往中下游流動，硝酸鹽因脫硝作用，濃度降低至檢測下限以下。淺層含水層受硝態氮污染已是全世界所面對的嚴肅課題，台灣亦然，並且台灣特殊的沖積扇含水層、其扇頂部份缺乏泥層保護，加上人口及農業活動密集，化肥或生活污水可能進入含水層。目前許多沖積扇地區，其地下水仍是自來水水源之一，當局應重視此區域性的污染。後續研究應擴大監測的樣本數，例如監測自來水水井及居民自有水井，以瞭解居民飲用水的硝態氮濃度。

關鍵詞：扇頂補注區、濁水溪、屏東平原、地下水、硝酸鹽

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前言

沖積扇地層廣泛分布於台灣島山脈的東西部兩側（林朝棨，1957），是重要的地下水資源區，其中以濁水溪沖積扇（或稱濁水扇洲）及屏東平原東側的沖積扇群最為重要，估計地下水年使用量約50-60億噸，占全台抽用量之80 %（經濟部水資源局，1999）。扇頂補注區的地層以礫石層為主，透水性佳但缺乏泥層隔絕，污染很容易隨地下水流動，在溶氧濃度高時，水中含氮的物質常被氧化為硝酸鹽，導致地下水中硝酸鹽濃度過高（陳文福等，2003）。含氮物質來源可能是農業化肥、生活污水、化糞池污水等。飲水中含過高硝酸鹽將對人體產生許多危害，例如：硝酸鹽在消化道中被還原成亞硝酸鹽，於高濃度（>10 mg/l）時，可能造成孕婦流產及新生兒藍嬰症缺氧致死；中濃度（1-10 mg/l）時、硝酸鹽為一氧化劑，可能會致癌（Follett, 1989），目前我國環保署的飲用水水源標準之硝態氮限制濃度為10 mg/l。

沖積扇地層的地下水，通常由扇頂進入、往扇中及扇尾流動。地層中有許多元素會參予氧化還原反應，例如：有機碳、溶氧、氮、硫、及重金屬（如鐵及錳），不但會形成新礦物並且也會改變地下水水質。氧化還原反應通常有微生物參予及催化，因反應所提供能量有高低差別，因此有些優先進行、有些較後（Chapelle, 2001）。以還原系列為例：溶氧還原優先進行、然後依次硝酸鹽還原、錳還原、鐵還原、硫酸鹽還原、最後甲烷生成（Stumm and Morgan, 1981）。一般而言，剛補注入地下的水含有溶氧，往深處流動時，因無空氣接觸，漸被地層及地下水中的耗氧物（例如有機碳）消耗，溶氧漸漸減少，水流越深或越下游（進入拘限水層），還原系列依次進行，地下水質因此會有帶狀特徵分布（Champ *et al*., 1979; Chapelle, 2001）。除了地下水環境外，海洋沉積物也有類似的現象，由海水-沉積物交界面往越深處，還原程度越增加，氧化還原帶狀特徵依次出現，稱之為「成岩序列」（diagenetic sequence）（Froelich *et al*., 1979）。

溶氧及硝酸鹽帶通常分布在地下水的源頭補注區，含溶氧帶的地下水，溶解之亞鐵離子較少，含氮物質（如有機氮及氨態氮）會傾向形成最終產物--硝酸鹽，以濁水溪沖積扇為例，扇頂的觀測井如二水、田中、六合等井，都發現有中高濃度（>1 mg/l）的硝態氮（圖1及圖2）。統計1956-2001年的數據顯示，田中地區地下水的硝態氮濃度，可能從1991年開始升高（謝永旭，1994-2003；陳文福等，2003）。1997-2001年監測扇頂34口井也發現地下水中硝態氮濃度界於0.47~17.8 mg/l，四年來呈穩定增加的趨勢，累積曲線中間值分別為：0.47、1.26、1.56、1.82 mg/l，年增加率約為0.28 mg/l（陳文福等，2003）。本文的目的，為整理及分析近十年來

（1997-2007）的地下水觀測網監測數據，探討濁水溪沖積扇及屏東平原的硝態氮濃度分布與趨勢，提供相關單位參考，呼籲國人重視此一污染問題，並提醒地區民眾應防範可能的健康威脅。

研究方法

本研究使用的井爲經濟部水利署及中央地質調查所設置的觀測井（經濟部中央地質調查所，1995，1999；台糖公司地下水中心，1999-2004）。濁水溪沖積扇的觀測站井共有191口，選擇其中位於扇頂補注區的22口，1997年至2001年爲每年監測一次，2002年至2003年、爲每月一次監測，2004至2005則監測20口（其中兩口因鄰近施工而廢棄）、旱雨季各一次，2007年的數據則是引用自農業工程研究中心（2007）的監測報告（表1、表2）。屏東平原的觀測井總數有134口，其中43口的硝態氮較高，應屬於扇頂補注區，1998-2007年每年監測一次（表3）。

採樣法依照環保署公告「地下水採樣方法」(NIEA W103.50B)，採樣前需預抽三倍井水(purging)。抽水時每隔5分鐘，記錄地下水之溫度、pH、導電度（EC）、水位、濁度，當抽出三倍井管水後、或pH及EC值穩定 (pH±0.1、EC±5 %)，才停止抽水與採樣，並填寫現場採樣記錄表。水樣以冰桶保存，於24小時內送回化驗室，硝態氮的檢測法爲馬錢子鹼比色法（NIEA W417.50A），檢測下限爲0.06 mg/l。

濁水溪沖積扇

彰雲地區含高濃度硝態氮的地下水主要分布於扇頂，氚定年數據顯示，都是50年內補入的新水（劉聰桂，1993-2001；Liu, 1995；陳文福等，2003），同時扇頂補注區的地下水也含有溶氧，含氮的化合物在溶氧的氧化下，轉變爲硝酸鹽。根據地下水勘測隊（1957）的調查，除兩口井外，本區硝態氮的背景濃度約在0.2 mg/l以下。又根據自來水公司的水質監測，約在1991年後，田中地區地下水中的硝態氮濃度有明顯增高的趨勢，由小於2.0增加爲6-10 mg/l之間。

沿地下水流動至中下游，硝態氮濃度很明顯減低，例如在田中－田尾之間，地下水中硝態氮由6.79-7.16 mg/l、驟降爲0.12 mg/l或更低，往更下游，濃度都在0.2 mg/l以下（圖2）。此現象應是地下水已漸進入硝酸鹽還原帶，因地層中有機質降解，致使水中硝酸鹽被脫硝作用還原成氮氣（Stumm and Morgan, 1995）：

$$5CH_2O + 4NO_3^- + 4H^+ = 5CO_2 + 2N_2 + 7H_2O$$

若以濃度0.5 mg/l爲界，高濃度硝態氮的地下水分布，在彰雲地區五個含水層中，以第一含水層的分布最廣，達全面積的1/8，顯示越淺層、越接近補注源的地下水中含硝態氮濃度越高（圖3）。

1997至2007年補注區20口觀測井，硝態氮濃度歷年變化如圖4。依不同水文地質區，可分成四個次分區：彰雲扇頂、山區、雲林扇頂及斗六沖積扇。

一、彰化扇頂區：有田中、溪州、田尾、柑園及二水，以田中的濃度最高，平均約5-8 mg/l，2003年1月至6月有明顯超過環保署飲用水源水質標準10 mg/l；其次爲柑園及二水，二水井曾在2000年時超過標準；溪州及田尾之硝態氮平均都在0.5 mg/l以下，可能因爲已經進入硝酸鹽還原帶。

二、山區井：有新光、坪頂、竹山、社寮，以新光及坪頂的濃度最高，平均約6-12 mg/l，有7次檢測值超過10 mg/l。新光及坪頂井的深度都已進入頭嵙山層的砂泥互層段，其中坪頂井的深度達186 m，含水層中泥層增多，對於地面水的補注應有阻絕的作用，另外其中的有機質應能還原硝酸鹽，但此兩口井的硝酸鹽濃度卻非常高，可能顯示該地之含水層已有破裂。此兩井位於八卦及斗六台地隆起之背斜軸，其含水層應存在破裂面，致使地面水能迅速進入較深的地下水層。

三、雲林扇頂區井：有六合、烏塗、刺桐、西螺及石榴，以六合及烏塗的濃度最高，平均約4-9 mg/l，接近10 mg/l，但仍沒有超過標準值。刺桐、西螺及石榴之硝態氮平均大都在0.5 mg/l以下，可能因爲已經進入硝酸鹽還原帶。

四、斗六沖積扇井：有崁腳、古坑、溫厝、東和，以古坑的濃度最高，平均約6-12 mg/l，有16次檢測值超過10 mg/l。其餘崁腳、溫厝及東和之硝態氮平均大都在5 mg/l以下（圖4）。

初步發現地下水的硝酸鹽氮濃度以旱季較高、雨季時較低，如古坑、新光、田中、坪頂等，例如在2002年8-9月時其地下水位最高、但硝態氮的濃度卻最低（<5 mg/l）；2003年4-7月時地下水位最低、但硝態氮的濃度有明顯的偏高（>10 mg/l）。可能因雨季時大量降雨，可沖稀地下水中硝酸鹽氮濃度，但旱季時，缺乏雨水補注，若有高濃度的化肥及灌溉水流入地下，則易形成較高硝酸鹽氮濃度的地下水（圖5）。

統計1997至2007年的監測數據顯示，補注區20口井硝態氮濃度最高可達23.2 mg/l（新光一2007年），中位數於2000年時最高、爲3.57 mg/l（圖6）。十年來曾超過10 mg/l的井有：東和、坪頂、社寮、古坑、二水、新光、田中等7口，其中又以新光及古坑井超過10 mg/l的次數最多，顯示其地下水受到硝態氮污染的情況最嚴重（表2）。

屏東平原

本區地層以多個沖積扇組成，地形由東向西降低，扇頂處以厚層礫石層爲主，分別向西、西北、西南扇狀展開，地層向西砂泥層厚度漸厚、層次漸多。地下水的流向，基本受到沖積扇地層控制，流向有兩個主要分量，由東向西（沖積扇）及由北向南（由陸至海），合成區域流向爲由東北向西南流（圖7）。

屏東地區深度約250公尺內的地質概況，其東北部地層以厚層礫石層爲主、分層較不明顯，主要的阻水層只有一至二層，地下水上下流通限制較少。西南部地層－港東至林園間、則分層明顯，大致有二層主要阻水層，地下水上下流通已受限制。若以深度作爲分層參考，第一含水層的深度約在0-60 m，第二含水層的深度約在60-120 m，第三含水層的深度約在120-250 m，第三含水層又可分爲兩段，但需注意，不同含水層分層深度隨區域而變（經濟部中央地質調查所，1997）。

屏東平原的東及北側爲沖積扇的扇頂位置，屬於區域性的地下水補注區。與濁水溪沖積扇相同，屏東地區地下水的硝態氮濃度也以補注區最高（圖8及圖9）。除少數例外、往下游及往地層深處，因硝酸鹽還原作用，使硝態氮的濃度降低。萬隆－林園剖面中的潮州及崁頂，其第三含水層上段，雖深度達150 m，且上覆有泥層，但硝態氮濃度達1.36-1.73 mg/l，顯然與大趨勢不合。其原因可能因局部地下水流在此層壓力極大，是本區的受壓水層，也是地方上常利用的自流井，可能因人爲使用，加速地下水的循環流動，地下水的補充速度大於脫硝作用，因此仍有硝酸鹽的存在（圖8）。

若以硝態氮0.5 mg/l爲界，圖9顯示屏東平原地下水硝態氮濃度的分層分布。以第一含水層的分布最廣，達平原的1/2面積，往第二含水層、其分布面積仍可達1/3，至第三含水層則分布較小、較局限，只有在隘寮溪及林邊溪兩沖積扇的扇頂部份。根據地下水工程處（1961）的調查，本區40年前地下水的硝態氮濃度約在0.2 mg/l以下，近年高濃度的硝酸態氮，應是人類社會經濟發展所致，例如農田化肥、生活污水及畜牧廢水。

經過1998-2007年度的監測，屏東平原觀測井總數134口中有43口井的硝態氮濃度經常大於0.5 mg/l，大都位於扇頂區（表3）。其中以建興一的濃度最高，達17.3 mg/l（2004年），其餘如泰山一井，也高達10.6 mg/l（2004年）。有四口井的濃度曾超過10 mg/l：萬巒一、建興一、泰山一及新庄一，其中又以建興一較嚴重，十年間有四年的數據是超過環保署飲用水限值。

1998-2007年間硝態氮的最高極大值發生於2004年（17.3 mg/l建興一），該年的降雨量較

少（約1100 mm），而2001年的硝態氮的極大值爲最低（3.69 mg/l建興一），該年的降雨量較多（約3100 mm），此兩年的年雨量相差達三倍，可能是影響地下水中硝態氮濃度的重要因素之一（圖10）。較高的降雨量應能沖淡地下水中的硝態氮濃度，例如本研究統計之高降雨年之硝態氮極大濃度爲17.3 mg/l、低降雨年之極大濃度爲3.69 mg/l，兩者差約四倍。

硝酸鹽濃度影響因素

自然界氮源大多來自生物分解，供需之間在生態系統中維持平衡，除了乾燥環境的地下水有自然硝酸鹽濃度富集的現象外（Stadler *et al*., 2008），未受污染的地下水中硝酸鹽並不高（Follett, 1989）。近代因化肥過量施用及人口增長、污水排放造成的多餘氮源，滲入地下後污染地下水，不論是先進國家（Ball and MacDonald, 2001; West, 2001; Brink *et al*., 2008; Panno *et al*., 2008）、或是開發中國家（Tang *et al*., 2004; Xu and Usher, 2006; Tirado, 2007; Fetouani *et al*., 2008; Hu *et al*., 2008; Ma *et al*., 2008; Umezawa *et al*., 2008），已成爲世界各國的普遍現象（Chern *et al*., 1999; Shomar *et al*., 2008; Mahvi *et al*., 2005; Majumder *et al*., 2008）。

在台灣的沖積扇地區，地下水中硝酸鹽主要的來源，應該也是農業化肥及生活污水，當滲入扇頂地下後，因地下水中仍有溶氧，不利於脫硝作用，故水中硝酸鹽能持續存在（圖11）。降雨及河川水的補注，因稀釋而使硝酸鹽濃度降低，但在乾季時、因土壤水份蒸發，造成淺層地下水的硝酸鹽濃度升高（Holz, 2008）。地下水往中下游流動，因地層中有機質降解、或還原性礦物（例如黃鐵礦）釋出電子，使得溶氧減少、然後發生脫硝作用，造成硝酸鹽濃度降低（Rivett *et al*., 2008; Schwientek *et al*., 2008; Thayalakumaran *et al*., 2008），最後、硝酸鹽被還原至極低濃度，通常都低於檢測下限（0.06 mg/l）以下。

結論與建議

濁水溪地區191口觀測井中約有20口（約10 %）的硝態氮濃度偏高，多位於扇頂補注區。1997至2007年數據顯示，曾超過10 mg/l的井有：東和、坪頂、社寮、古坑、二水、新光、田中等7口，其中又以新光及古坑井超過10 mg/l的次數最多，硝態氮濃度最高達23.2 mg/l（新光一2007年）。

屏東平原134口觀測井中約有43口（約32 %）的硝態氮濃度偏高，其井數較多的原因爲屏東平原扇頂礫石層的分布比濁水溪地區更廣。監測十年間（1998-2007年）有四口井的濃度曾

超過10 mg/l：萬巒一、建興一、泰山一及新庄一，其中又以建興一較嚴重，十年間有四年的數據超過環保署飲用水限值，硝態氮濃度最高達17.3 mg/l（2004年）。

硝酸鹽主要的來源，應該是農業化肥及生活污水，降雨及河川水的補注，因稀釋而使硝酸鹽濃度降低，但在乾季時、因土壤水份蒸發，造成淺層地下水的硝酸鹽濃度升高。地下水往中下游流動，硝酸鹽因脫硝作用，濃度降低至檢測下限以下。

淺層含水層受硝態氮污染已是全世界所面對的嚴肅課題，台灣亦然，並且台灣特殊的沖積扇含水層、其扇頂部份缺乏泥層保護，加上人口及農業活動都非常密集，因此化肥或生活污水可能進入含水層。目前濁水溪沖積扇及屏東平原沖積扇，其地下水都還是當地自來水的水源之一，相關當局應重視此區域性的污染問題。後續研究應擴大監測的樣本數，為求監測水樣能代表居民的飲用水水質，應監測自來水水井及居民自有的水井，以瞭解居民目前飲用水的硝態氮濃度。

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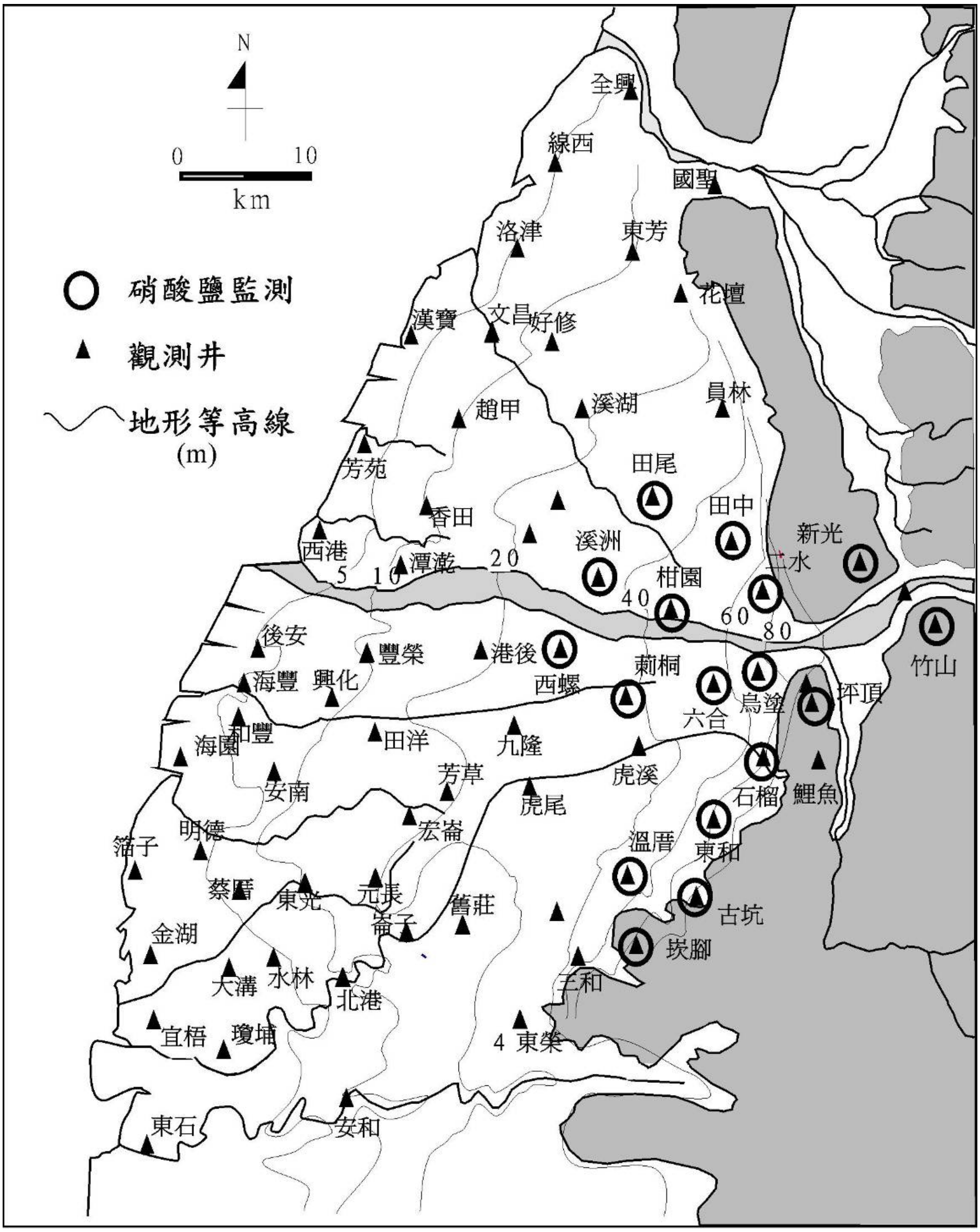

圖 1. 濁水溪地區地下水觀測站位置圖

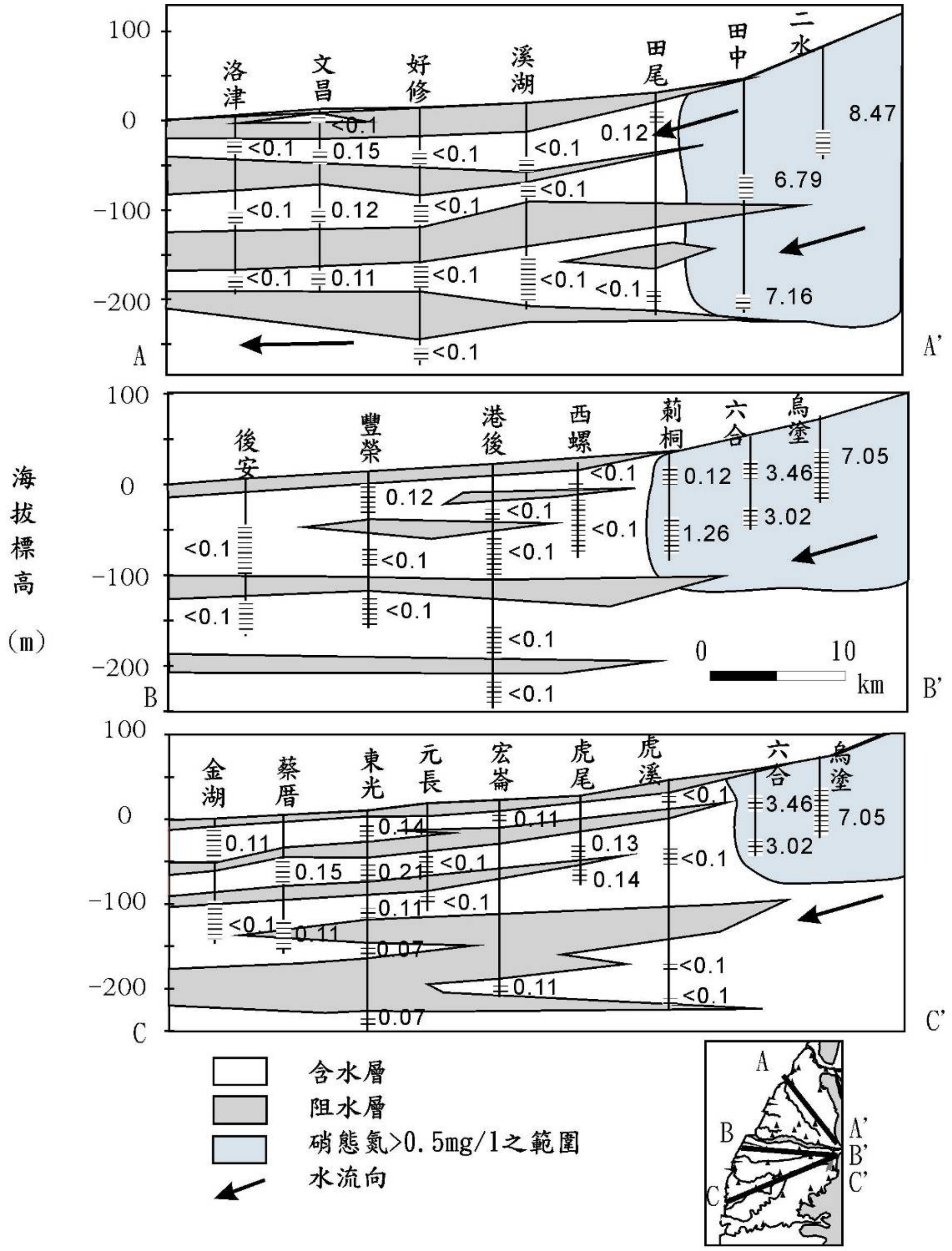

圖 2. 濁水溪沖積扇地下水之硝態氮濃度（2002 年）分布剖面

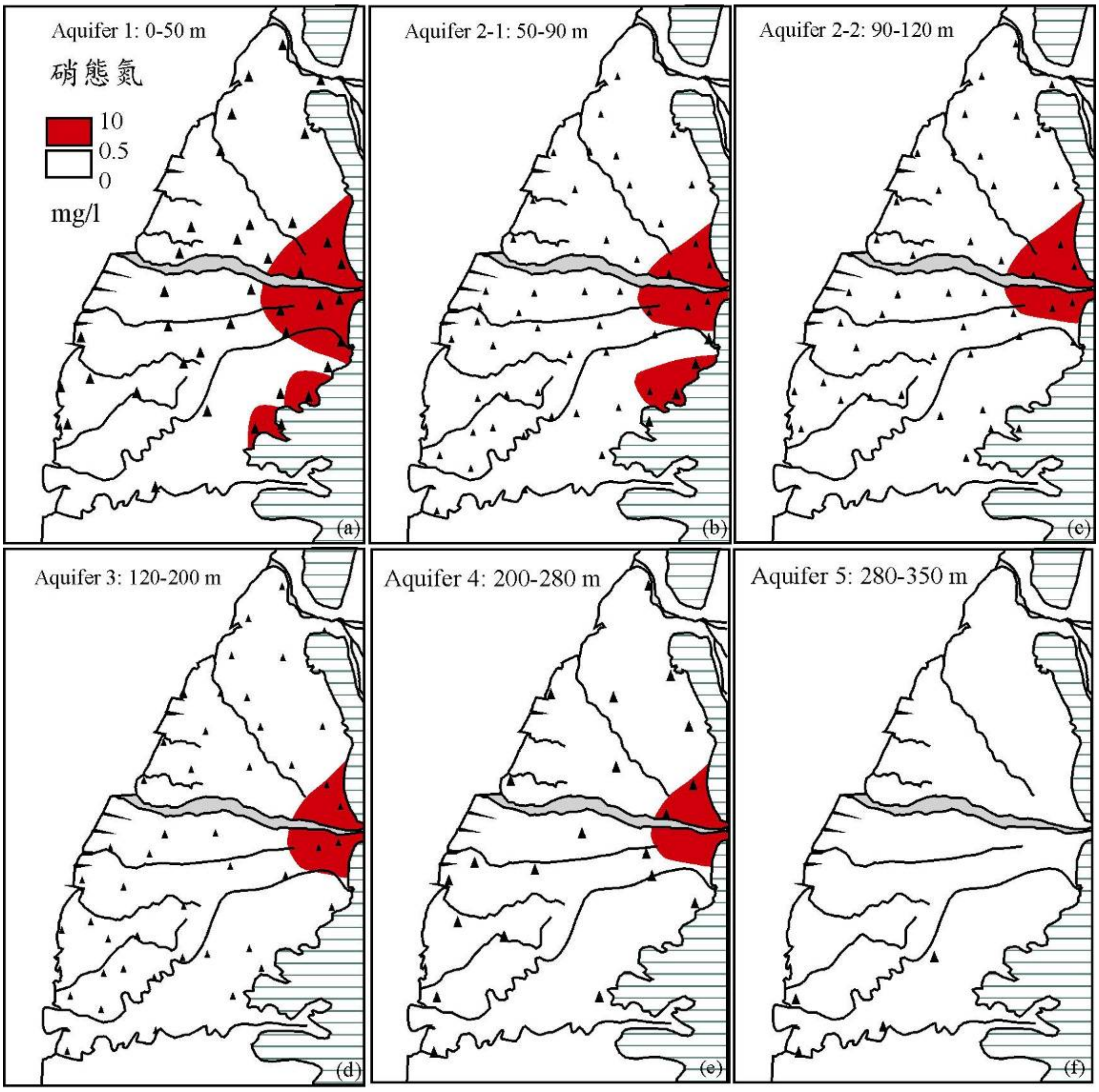

圖 3. 濁水溪地區地下水之硝態氮濃度大於 0.5mg/l 之分布

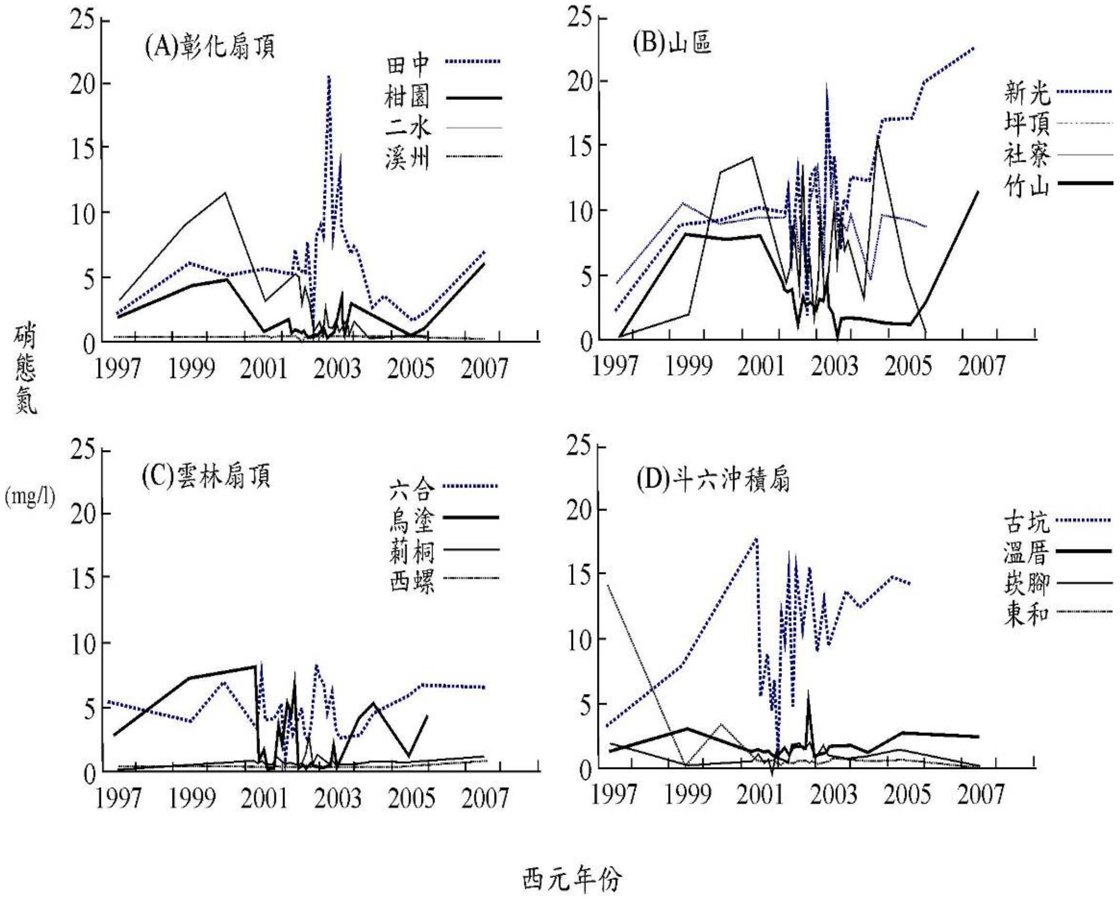

圖 4. 濁水溪補注區地下水中硝態氮濃度之監測結果

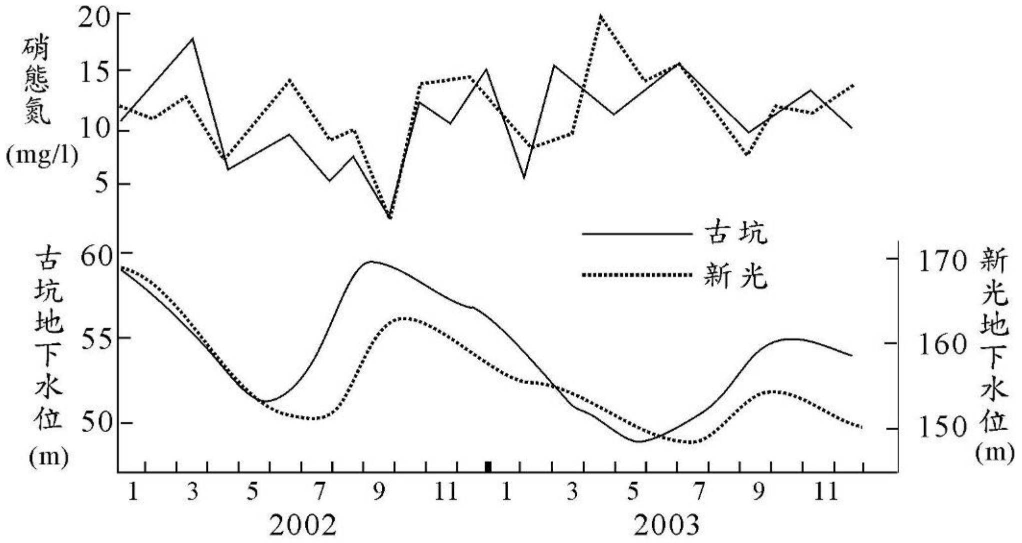

圖 5. 古坑及新光井硝態氮濃度與地下水位

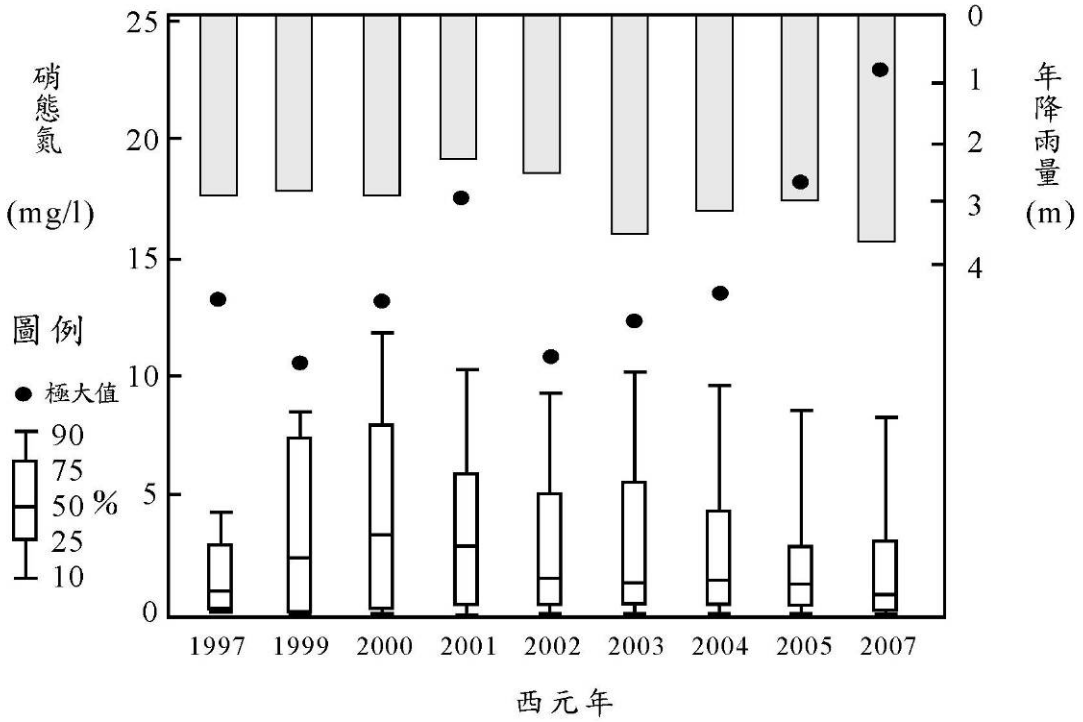

圖 6. 濁水溪補注區 1997-2007 地下水中硝態氮之濃度統計

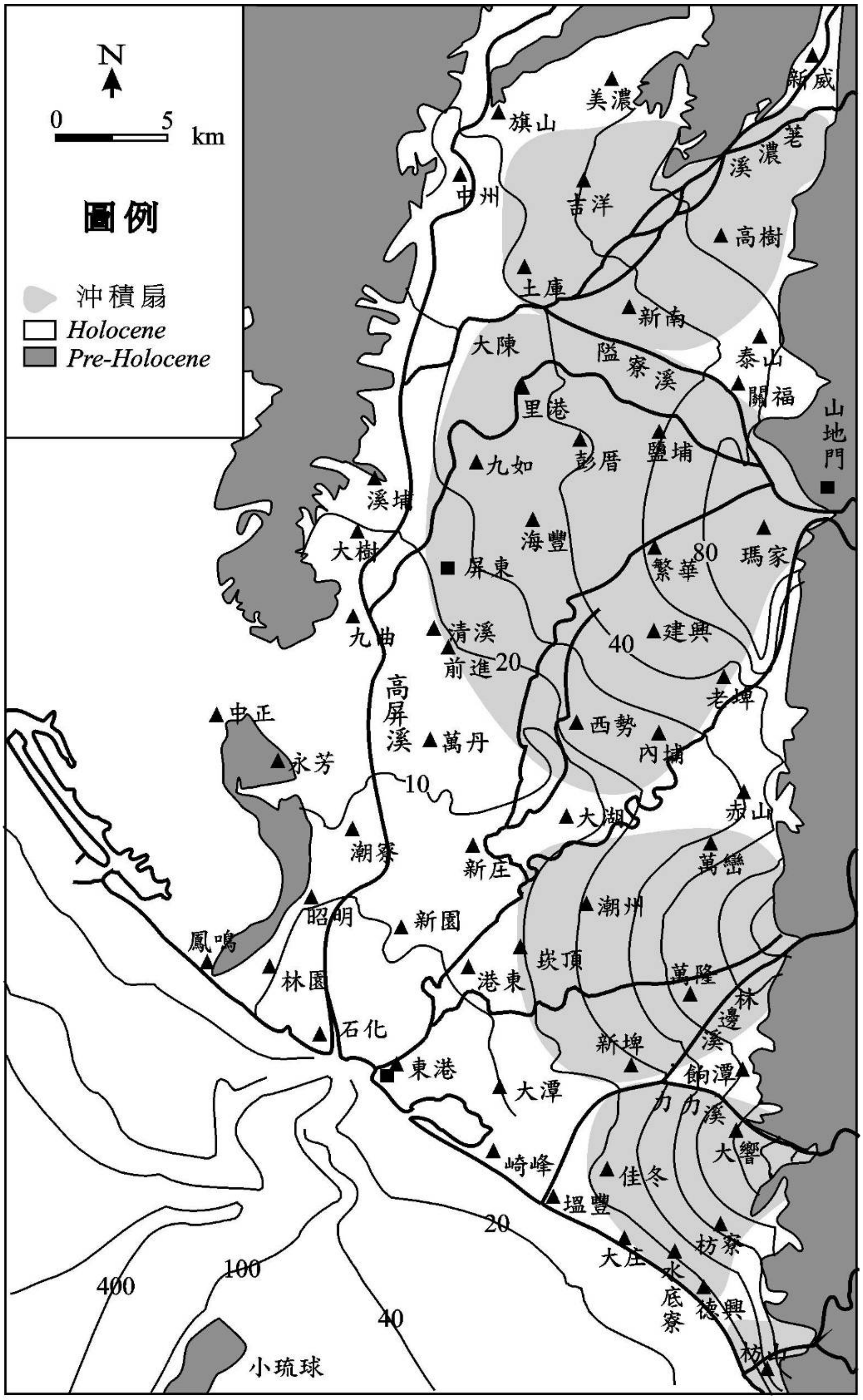

圖 7. 屏東平原之沖積扇分布與地下水觀測井位置

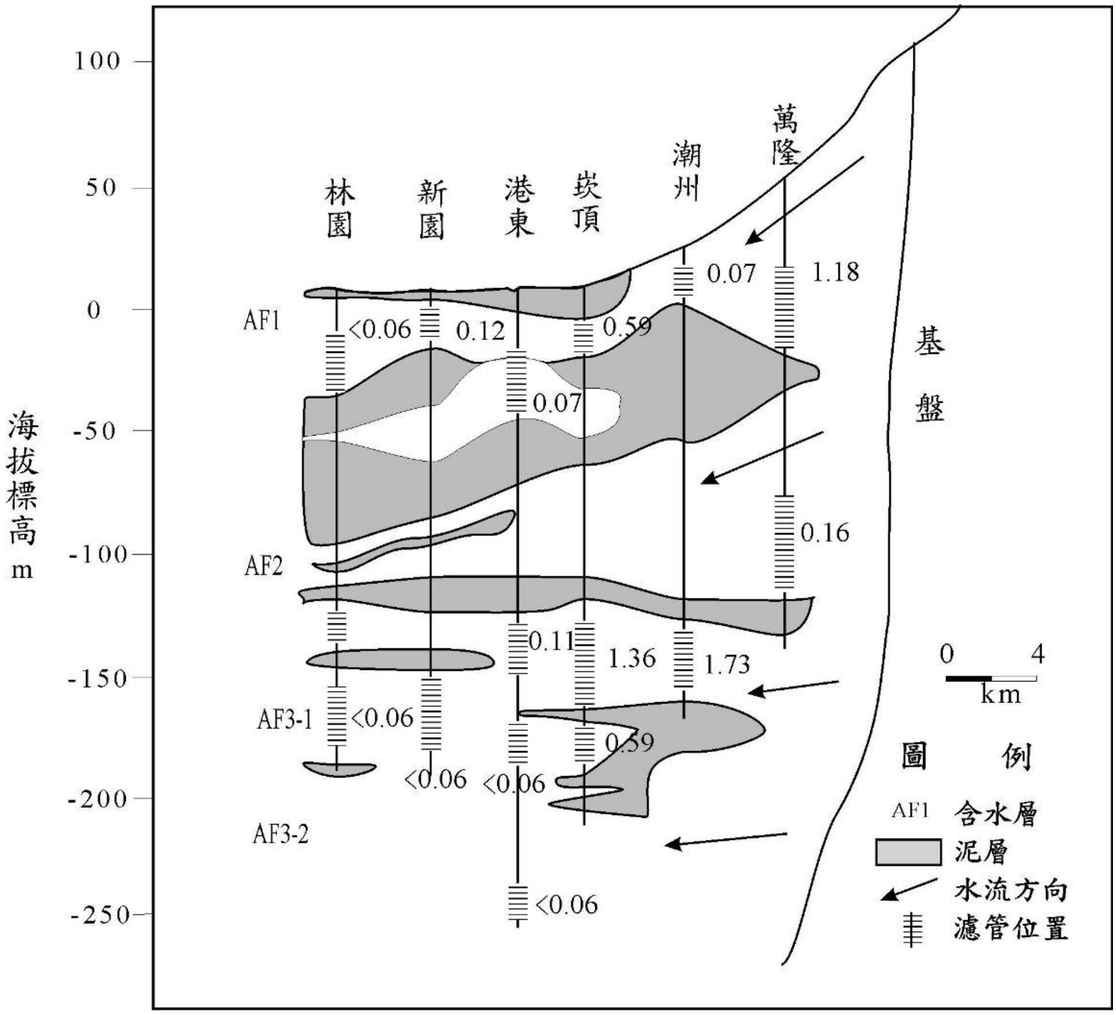

圖 8. 屏東平原地下水之硝態氮濃度（2005 年）分布剖面

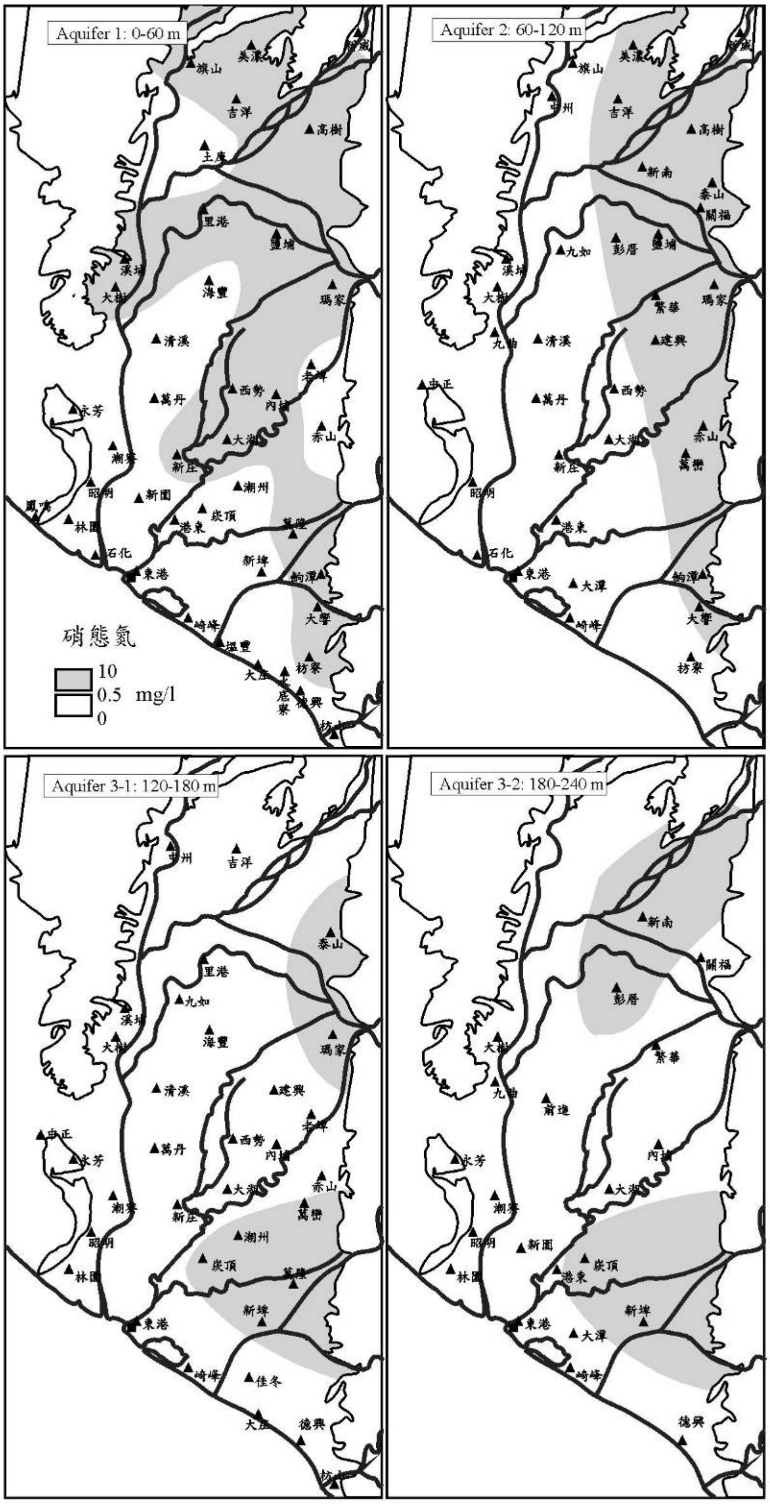

圖 9. 屏東平原地下水硝態氮濃度大於 0.5mg/l 之分布

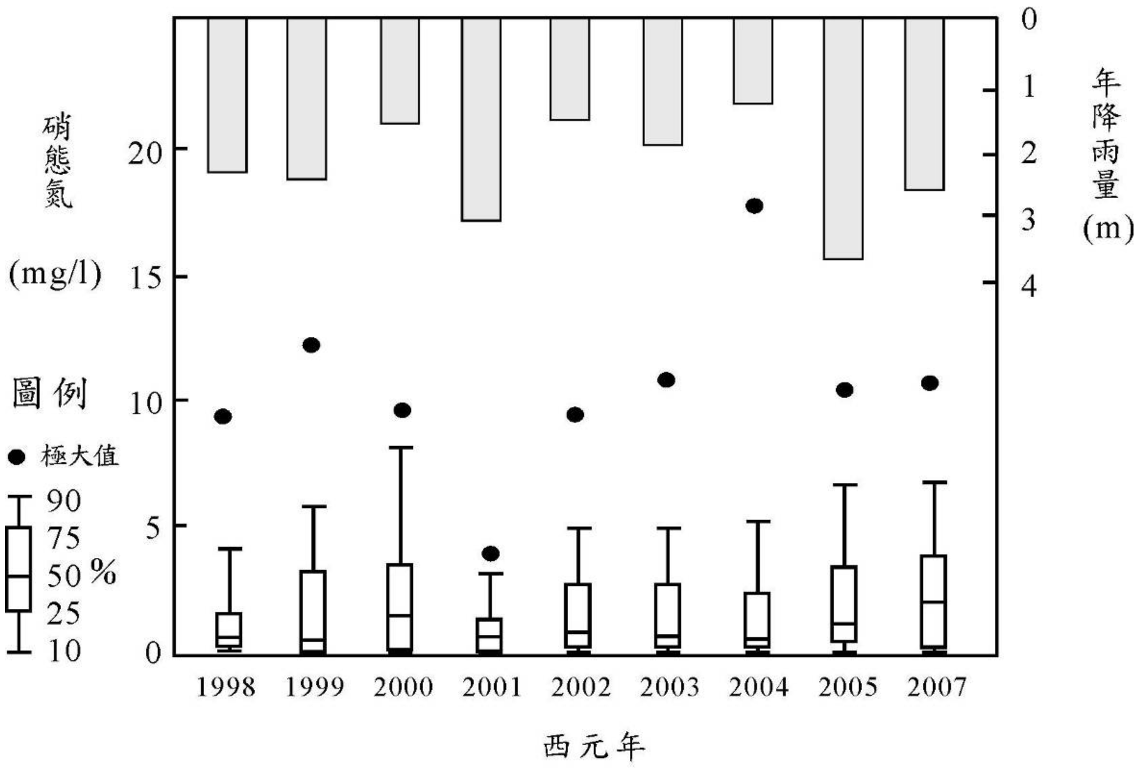

圖 10. 屏東平原 1998-2007 地下水中硝態氮之濃度統計

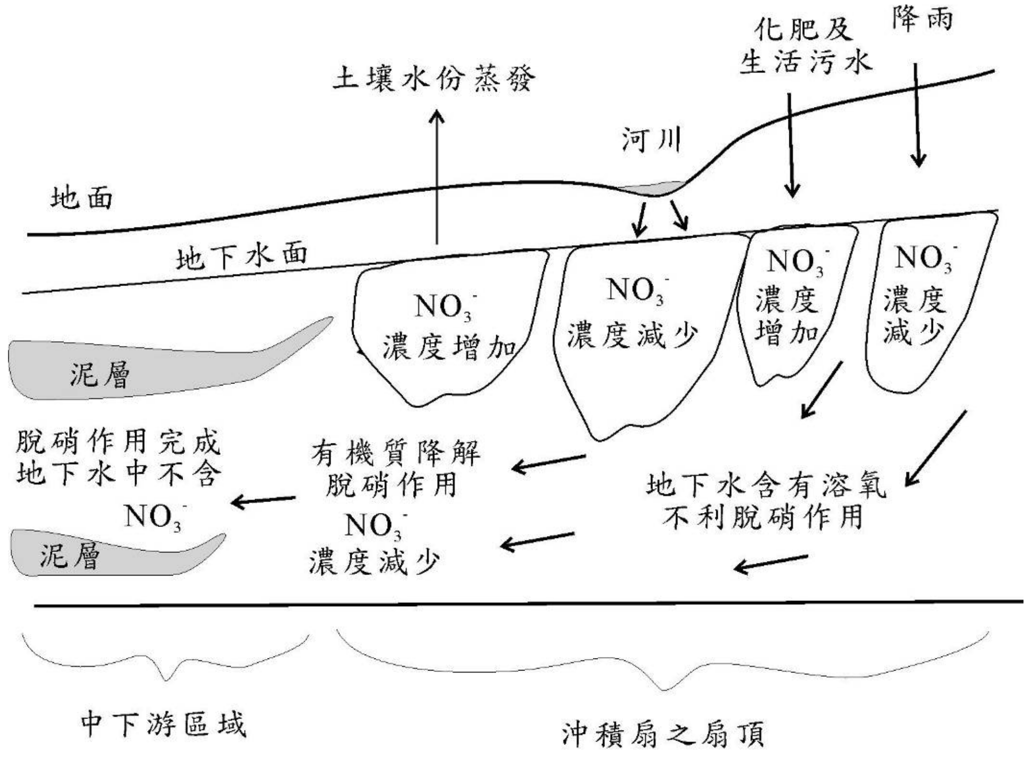

圖 11. 扇頂補注區地下水之硝酸鹽濃度影響因素

表 1. 濁水溪扇頂補注區觀測井之基本資料

井名	濾管起 m	濾管止 m	靜水位 m	X(m)	Y(m)	地表高程(m)
崁腳一	78	102	53.62	202021	2612378	110.11
古坑一	66	102	47.52	204980	2615932	100.67
溫厝一	14	32	7.54	199400	2617408	48.65
溫厝二	78	96	12.74	199400	2617408	48.65
東和一	21	57	14.43	205251	2620504	75.66
石榴一	9	15	7.37	206944	2624360	66.57
坪頂一	144	186	95.77	212545	2628234	297.60
烏塗一	40	94	29.26	208141	2629816	72.31
六合一	24	54	17.21	204584	2629706	56.18
莿桐一	14	32	6.45	197886	2628381	37.62
西螺一	20	38	4.82	194891	2632723	29.49
社寮一	9	18	9.16	220800	2634163	176.07
竹山一	66	96	6.36	217282	2629020	150.95
新光一	78	120	53.10	215584	2638740	222.11
二水一	72	108	39.15	210242	2634428	82.44
柑園一	14	32	9.40	201662	2635696	46.69
溪洲一	4	28	4.56	198371	2639267	34.18
田中一	98	134	11.48	207088	2639188	49.56
田中二	243	266	14.29	207088	2639188	49.56
田尾一	12	30	7.07	201045	2643280	32.55

靜水位指井口下之深度。座標爲二度分帶。

表 2. 濁水溪扇頂補注區之歷年硝態氮濃度（NO_3-N mg/l）

井名	1997	1999	2000	2001	2002	2003	2004	2005	2007
崁腳一	2.45	0.66	0.38	1.18	0.37	0.72	0.77	1.73	0.54
古坑一	3.15	7.41	12.20	17.80	10.46	12.63	13.70	14.77	
溫厝一	0.18	0.24	0.20	0.23	0.31	0.36	0.15	0.23	0.11
溫厝二	1.14	3.10	2.32	1.82	1.57	2.43	1.69	2.54	2.91
東和一	13.76	0.10	3.57	3.91	0.34	0.23	0.34	0.29	0.10
石榴一	0.08	0.14	0.26	0.05	0.07	0.12	0.10	0.12	0.24
坪頂一	4.47	10.40	9.14	9.77	9.10	9.88	7.10	8.74	
烏塗一	2.12	7.05	6.87	4.01	2.84	0.31	3.90	2.36	
六合一	6.46	3.46	7.99	4.15	5.02	5.82	4.58	8.52	8.67
莿桐一	0.02	0.10	0.13	1.27	0.06	0.03	0.10	0.11	1.92
西螺一	0.74	0.10	0.17	0.57	0.09	0.26	0.10	0.12	0.17
社寮一	0.47	2.13	13.40	14.10	6.74	6.4	9.33	2.36	
竹山一	0.02	7.24	7.05	7.81	3.09	2.29	1.44	2.27	9.77
新光一	1.55	8.99	9.94	10.20	9.77	11.35	14.50	18.59	23.20
二水一	3.65	8.47	12.70	3.32	3.21	1.24	0.10	0.33	
柑園一	1.99	4.80	5.30	0.72	0.59	1.35	2.37	0.59	6.07
溪洲一	0.16	0.26	0.18	0.66	0.14	0.11	0.10	0.10	0.04
田中一	2.36	6.79	5.75	5.96	6.61	10.47	3.29	2.08	7.65
田中二	0.06	7.16	2.95	2.95	2.31	5.18	3.09	5.02	8.02
田尾一	0.10	0.12	0.14	0.10	0.37	0.15	0.10	0.10	0.03

1. 2003-2004 年數據爲每月加總之年平均值。
2. 2007 年數據引自農業工程研究中心（2007）

表 3. 屏東平原扇頂補注區之歷年硝態氮濃度（NO_3-N mg/l）

井名	1998	1999	2000	2001	2002	2003	2004	2005	2007
萬巒一	9.50	8.41	9.45	2.85	3.92	8.31	9.80	9.89	10.50
建興一	8.41	12.50	9.64	3.69	5.96	10.4	17.30	10.43	
泰山一	9.59	7.48	9.20	2.69	9.47	7.36	10.60	9.77	9.22
溪埔一	8.28	7.82		2.11	7.30	5.67	8.85	8.05	
鹽埔二	7.90	5.98		1.30	2.34	1.37	2.88	3.07	3.30
吉洋		8.41	7.42	2.14	3.80	5.94	5.23	3.93	4.30
彭厝一	3.57		4.12	1.90	2.61	4.82	6.08	5.41	4.58
鹽埔一	5.50	1.87	2.23	1.00	1.50	0.11	1.52	1.64	0.80
旗山一	0.11		3.70	0.37	5.43	7.53	5.63	5.20	0.89
繁華一	2.75		6.82	1.93	4.33	3.25	3.38	5.09	6.43
崁頂一	3.10	0.74	0.39	0.20	ND	ND	0.62	0.59	
內埔一	6.92	4.57	9.23	2.82	1.40	0.75	1.42		1.66
枋寮一	4.46	3.68	5.76	1.19	0.45	2.28	2.93	3.64	4.96
高樹一	1.92	2.39	2.72	1.88	1.61	4.53	2.85	5.57	2.38
新南一	0.27		5.38	1.13	2.18	3.49	4.05	3.55	
瑪家一	ND	2.39	3.05	1.43	1.44	1.80	3.10	4.57	
建興二	2.51	0.38			2.67	3.22	2.67	2.57	
餉潭一	ND		3.56	0.96	1.76	0.48	2.49	3.57	
新庄一	0.33	0.63	0.32	1.11	0.66	ND	1.22	10.48	
高樹二	1.95	1.84		0.89	2.15	1.96	2.96	2.86	1.69
海豐二	8.63	0.84		0.55	0.83	1.04	1.29	1.41	1.68
大樹一	0.25		1.11	1.25	5.85	3.76	1.43	0.89	
大嚮一	2.84	0.61	4.85	0.30	2.72	0.23	4.30	0.75	
彭厝二	1.48		1.65	0.77	2.39	2.07	2.75	2.75	3.99
大湖一	0.24		0.55	0.94	2.39	2.10	3.15	4.34	
美濃二	0.17	2.82		1.36	1.67	2.63	2.32	2.34	5.40

續表 3. 屏東平原扇頂補注區之歷年硝態氮濃度（NO_3-N mg/l）

井名	1998	1999	2000	2001	2002	2003	2004	2005	2007
萬隆一	2.71	3.02	2.28	0.59	1.95	1.16	1.39	1.18	
西勢一	0.17	2.62	2.15	0.72	3.22	0.40	2.36	1.43	
瑪家二	1.12	2.59		0.83	2.09	0.16	1.69	1.98	2.57
潮州二	ND	1.55		0.54	1.36	ND	1.94	1.73	1.76
吉洋一	0.30		2.41	0.85	1.09	1.81	1.82	1.61	4.30
新南二	0.30		1.51	0.61	2.65	1.56	1.50	1.41	
泰山二	3.47	1.23		0.44	0.98	0.73	0.85	1.07	1.54
里港一	0.74	1.31	ND	0.67	1.54	1.56	1.78	0.59	2.13
新威一	0.49		0.26			ND	1.34	1.70	7.06
關福一	0.15		1.37	0.67	0.67	1.24	1.28	1.00	
萬隆二	0.36	0.23			ND	2.95	0.26	0.16	
大潭二	0.63	0.36	0.20			0.15	2.08	0.68	
大嚮二	0.74	0.78		0.36	0.99	0.26	0.56	0.70	
新埤二	0.62	0.62		0.23	ND	ND	0.87	0.75	1.22
瑪家三	0.16	0.91		0.19	ND	ND	0.30	1.27	1.47
繁華二	0.85		0.55	0.23	0.54	0.40	0.66	0.70	0.72
中州一	0.30	0.28				ND	0.80	0.86	

ND 爲小於 0.06 mg/

Nitrogen and Phosphorus Contamination of Groundwater in Some Agricultural Area of Taiwan

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Abstract

To understand the current status of nitrogen and phosphorus contamination of groundwater resulting from fertilization on farmland, the history data from monitoring network by Water Resources Agency (WRA) and Environmental Protection Administration (EPA) have been analyzed and with groundwater quality on some agricultural regions sampled and measured. The results reveal quiet high percentages of monitoring wells set by WRA and EPA with NH_4^+-N contents in groundwater exceeding drinking water quality standard (0.1 mg/l), even higher than the limit of the second type groundwater pollution monitoring standard (0.25 mg/l). Based on the investigation of agricultural regions, the NH_4^+-N contamination from agricultural area seems similar as that from different land uses monitored by WRA or EPA. There are less 10 % of monitoring wells by WRA or EPA or sampled from agricultural regions of which NO_3-N contents are higher than the limit of drinking water and exceeding the limit of drinking water quality standard (10 mg/l). However, no data higher than the limit of the second type groundwater pollution monitoring standard (25 mg/l) are observed. There is no significant correlation between the N content in groundwater and soil profile texture. However, the N contamination has been getting gradually higher on some agricultural region located in head recharge area of Choshui alluvial fan with few mud layers in the geological profile, even the soil texture is clay. One of the reasons might be attributed to heavy rate of fertilization by local farmers and need to be further confirmed in the future. The results also indicate almost the total P is much lower than limit of the groundwater quality standards (0.15 mg/l) stipulated by the Netherland.

Keywords: Nitrogen, Phosphorus, Groundwater, Agriculture

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Introduction

The contamination of the groundwater resulting from agriculture fertilization is getting more concerned than that from industry, recently. According to an investigation, the nitrogen efficiency is lower than 50 % in paddy field, much less in upland than in paddy field and only 11 % in vegetable land in Taiwan (Lian *et al*., 1996; Lian and Wang, 1997). The improper fertilizer use will result in the nutrient loss and emission to the environment. When nutrients input exceeding over the amount that can be uptake by crop, it might leach into and contaminate the groundwater rendering hazards to human health (Lal and Stewart, 1994). The nutrient loss via the runoff to surface water might cause eutrophication and through denitrification might emission to become the greenhouse gases (Forman *et al*., 1985；Addiscott *et al*., 1991; Janzen *et al*., 2003).

The nitrate concentration of groundwater under agricultural land was becoming higher and higher in some area of Taiwan according to reports of some studies. Furthermore, 75 % of tap water sources from groundwater in Taiwan, yet, 30 % of that have already been contaminated to different extent. The nitrogen contamination of groundwater is not only from the application of chemical fertilizer but organic fertilizer (Bruno and Ritchie, 2005).

There are two networks of quality monitoring for groundwater in Taiwan. One is the environmental water quality monitoring and information system build by the Environmental Protection Administration (EPA), Executive Yuan, the other is groundwater monitoring network build by the Water Resources Agency (WRA), the ministry of economic affairs. However, Due to the establishment of the monitoring networks with their specific purposes, the monitor data might not be easy apprehensive for the contamination status of groundwater from agriculture.

To access the present status of groundwater quality influenced by fertilization at agricultural region, not only the history data of networks mentioned above have been reviewed, but also conducted the investigation for the groundwater in some agricultural region in this study.

Material and Methods

The sampling and monitoring of two reviewed networks and the investigation of groundwater at some agricultural regions were described as the followings:

(1) Groundwater monitoring network by WRA

The preliminary monitoring groundwater started from 1958 by the water resources agency (WRA). However, the whole monitoring network was established until 1995. The wells were installed with depth 25-300 m to monitor the lower aquifer. Groundwater samples were regularly

collected and analyzed. The frequency of the monitoring is once half a year. Some monitoring has been conducted for the understanding for the hydrology of geology, and monitoring for groundwater resource and quality for important groundwater region then gradually conducted. Some stations were established specifically for the monitoring and study for the subsidence near seashore in southwestern Taiwan. The purpose of the network is to realize the environment of groundwater, including the geographical and hydrological characteristics, water level and quality, and establish the database to provide the related information based on the spatial and temporal change of groundwater amount and quality for decision making and strategy analysis. The data for analysis in this study were retrieved from network of WRA.

(2) Network of groundwater quality monitoring by the EPA

The long term monitoring for groundwater has started from 1995 by the EPA. Most of the wells were installed with depth 10-20 m to monitor the uppermost aquifer. Groundwater samples were regularly collected and analyzed. The frequency of the monitoring is once each season. The purpose of the network is to establish the long term database and maintain the national source of the groundwater and prevent the contamination from point pollution.

Since each monitoring network has their specific purposes, their sampling sites, depth, frequency, interval are different. The differences between the networks are as the Table 1. Items of water quality measured by both monitoring network are similar, including the NH_4^+-N, NO_3-N. As for maintaining convenience, the monitoring wells of both networks were installed at school or public organization, hence, monitoring sites might far from agricultural region. The data for analysis in this study were retrieved from network of EPA.

(3) Investigation for the quality of groundwater at some agricultural region

To realize the present status of groundwater quality under agricultural region, the main agricultural region in middle and south Taiwan are planed to be investigated. However, the project have not yet finished up to this day, only the whole Changhwa county and part of Yunlin and Pingtung county have been investigated from May to Oct. ,2008. The sampling period for Chunghwa area is from 04/03/2008 to 06/02/2008，for Yunlin area from 06/02/2008 to 06/11/2008, for Pingtong area from 09/09/2008 to 10/22/2008.

There are 5 to 30 groundwater samples per town at agricultural region. The groundwater was sampled randomly from farmers’ wells for irrigation. The sampling sites were located by GPS. Most of the wells are shallow well about 20-30 m in depth. The sampled groundwater are kept in 4 ^{o}C refrigerator from sampling site to lab and measured by the following methods regulated by EPA:

1. pH: Electrode Method (EPA, 2004)
2. EC: Electrical Conductivity Method (EPA, 2000).
3. NH_4^+-N: Indophenol Colorimetric Method (EPA, 2005)
4. NO_3-N: Cadmium Reduction Method (EPA, 2000).
5. Total P: ICP-AES Method (EPA, 2004)
6. SO_4^{-2}: Nephelometry Method (EPA, 2000)
7. CL^-: Mercuric Nitrate Method (EPA, 2002)

Result and Discussion

To assess the quality status of groundwater, the data were compared to “drinking water quality standards” and “The second type groundwater pollution monitoring standards”. The second type groundwater is defined as groundwater outside the water sources and quality protection area for drinking water. The NH_4^+-N limits for each standard are 0.25 ppm and 0.1 ppm, respectively. The NO_3-N limits for each standard are 10 ppm and 25 ppm, respectively. The NH_4^+-N standard of the second type groundwater falls between the standards of UK and US (Table 2). The NO_3-N standards of UK, US and the Netherland are about 10 mg/l. It is similar with the standard for drinking water. The NO_3-N standard of the second type groundwater is much higher than that of other countries. Most countries have no standard for phosphorous in groundwater except the Netherland. The limit for phosphorous in groundwater is stipulated 0.15 mg/l by the Netherland.

The limit of Nitrate-N content for drinking water in many countries, such as UK, US, Germany and WHO, is about 10 mg/l the same as that in Taiwan.

Based on results of the team of groundwater investigation on 61 wells in Chunghwa and Yunlin area in1956, the nitrate content in groundwater was 0.02-1.8 mg/l, the mean is was about 0.16 mg/l (Team of groundwater investigation, 1957). These data are the lost early data among the investigation. They could serve as background of the groundwater that have not yet contaminated.

(1) Groundwater monitoring network by WRA

The NH_4^+-N and NO_3-N contents in groundwater data are recorded in database of network in 2001. The phosphorous contents have not been measured until now. The number of total monitoring wells and percentage of wells with NH_4^+-N and NO_3-N content higher than limit of drinking water quality standards and the second type groundwater pollution monitoring standards are listed in the table 3 and Table 4, respectively.

The percentages of wells with NH_4^+-N content higher than limit of drinking water quality

standards and the second type groundwater pollution monitoring standards are higher and higher from 2000 to 2007. The NH_4^+-N contamination becomes worse without prevention control, even the depths of well are quiet deep about 25-300 m. Unfortunately, the background situation has not yet founded to compare the current status.

As to the nitrate-N contents, < 1 % of total monitoring wells are higher than the limit of the drinking water quality standards (10 mg/l). The results reveal current status of NO_3-N contamination is not as serious as NH_4^+-N contamination; even though nitrate is easier to leaching than ammonium through soil profile.

(2) Network of groundwater quality monitoring by the EPA

The NH_4^+-N and NO_3-N contents in groundwater data are recorded in database of network in 2000. The phosphorous contents have not been measured until now. The number of total monitoring wells and percentage of wells with NH_4^+-N and NO_3-N content higher than limit of drinking water quality standards and the second type groundwater pollution monitoring standards are listed in the table 5 and Table 6, respectively.

The percentages of wells with NH_4^+-N content higher than limit of drinking water quality standards and the second type groundwater pollution monitoring standards are quite high from 2000 to 2007. It seems the contamination is very serious, especially in Taipei, Tainan, Kaohsiung, Pingtung and Changhwa counties. However, the percentages are become lower gradually and the contamination is well controlled gradually.

The Change of NH_4+-N contents in some monitoring with higher contents are as the fig.1. The highest one is found at Tongshan element school in Ilan. The high NH_4^+-N contents maintained between 50-70 mg/l from 2000 to 2007. As is located between the downtown and agricultural area, the NH_4^+-N contamination might be caused by both of communities and agriculture. The high NH_4^+-N content may be due to very shallow groundwater level about 1.8-2 m. Another two high contents are contaminated from communities in Taipei and Hsinchu city. The groundwater level of the monitoring well at Hsinchu city is 80-90 cm. The NH_4^+-N contamination is apparently from community. The NH_4^+-N contamination is reduced in Taipei city gradually, but not reduced in Hsinchu city. One of the monitoring well is located in agricultural area at Chiku town, Tainan County. The groundwater level is about 2 m. The NH_4^+-N contamination is suggested from agricultural practice and becomes higher and higher gradually.

As to the nitrate-N contents, about 1-2 % of total monitoring wells are higher than the limit of the second type groundwater pollution monitoring standards (25 mg/l). While 8 % of wells are

higher than the limit of the drinking water quality standards (10 mg/l). As the results of monitoring by WRA, it reveals current status of NO_3-N contamination is not as serious as NH_4^+-N contamination; even though nitrate is easier to leaching than ammonium through soil profile.

(3) Current status of groundwater quality at some agricultural region

There are 440 groundwater samples already collected until now. The ranges of items measured for groundwater samples are listed in Table 7. The range is quite wide for the EC, SO_4^{-2}, Cl^- content in groundwater.

There is no standard for the phosphorous content in the groundwater and drinking water in Taiwan and many countries. However, that of groundwater is stipulated as 0.15 mg/l in the Netherland. There are only 3 data sampled from coast area of Chunghwa County with P content higher than 0.15 mg/l. The highest P content around 6.3 mg/l is much higher than the limit of standard. Sustained efforts should be continually made to identify the source of the contamination and prevent the contamination in the future. Another higher P content about 0.48 and 0.18 mg/l are slightly higher than the limit of standard. The rest data are under the limit range. Based on the distribution of P in monitoring, apparently, the P contaminations only appear at few sites in agricultural land.

About 52 % of NH_4^+-N contents of groundwater samples exceeded the limit (0.25 mg/l) of the second type groundwater pollution monitoring standards. About 70 % of those exceeded the criteria of the drinking water quality standard (0.1 ppm) based on the analysis on the total of sampled data (Table 8). The NH_4^+-N contamination from agricultural area seems similar as that from different land uses monitored by WRA or EPA. While compared to that of groundwater quality monitoring network by EPA in 2007, the NH_4^+-N contamination from agricultural area seems more serious than the data from other different land uses.

The NH_4^+-N contaminations in Changhua and Yunlin counties are more serious than that in Pingtung area (Fig. 2,4,6). The highest value in Chunghwa County is 7.5 mg/l, approximately 30 times the limit. All of the samples collected from Shengang, Siansi, Yuanlin, Dacun and Pusin Township are higher than 0.25 mg/l. The highest value in Chunghwa County is 8.3 mg/l, approximately 34 times the limit.

The NO_3-N concentrations of all samples are lower than the criteria of the second type groundwater pollution monitoring standards (25 ppm). However, there are about 8 % of samples exceeding the limit of drinking water quality standards (10 ppm) (Table 7). Compared to that of groundwater quality monitoring network by EPA in 2007, the NO_3-N contamination from

agricultural area seems similar with the data from other different land uses.

There are 278 samples collected in Changhua area and some groundwater has been collected at Mingchien town by another project in 2002 (Fig. 3). Some of the investigated wells with NO_3-N contents are higher than the limit of drinking water quality standards (10 mg/l). For examples, two samples from Fangwan Township are are 19.59 and 20.74 mg/l, respectively. Besides, the samples from Tienchung, Chichou, Changhwa and Mingchien, Nantou that located in head recharge area of Choshui alluvial fan. There are few mud layers in the geological profile, so the nitrate leaching is abundant (Chen, *et al*., 2003). The N contamination has become higher and higher gradually in that region.

There are 41 samples collected in Yunlin area (Fig. 5). The nitrate-N contents of two samples are higher than the limit of drinking water quality standards (10 mg/l) at Tzutung and Kukeng Townships, respectively. The area with higher NO_3-N contents seems also located the head recharge area of alluvial fan.

Among all of the samples, the area with higher NO_3-N contents is located in Pingtung county (Fig. 7), and about 21 % of wells of total 121wells sampled with NO_3-N are higher than the limit of drinking water quality standard. (Fig. 9). The higher NO_3-N contents are distributed at Kaoshu, Yenpu and Chungtzu townships especially. The NO_3-N leaching situation are very obvious at Changtzu Township, The nitrate-N concentrations of 12 samples of total 21 samples are higher than the quality of drinking water standards (10 mg/l).

Due to texture of the heavy soil with higher water holding capacity than light soil, the nitrogen leaching can be expected to be more serious in sandy soil than in heavy soil (Lian *et al*., 1996). Based on the study in lysimeter, The nitrogen leaching to one meter below soil were < 1 % of applied rate during the growth stage in medium and fine texture soil. While the nitrogen leaching might be reach 20 % of applied rate if growth of vegetable was not well in sandy soil. (Chen *et al*., 2006).

For better understanding the correlation between the nitrogen leaching and soil texture, 5 classes for soil profile texture of soil management group in soil database established by the agricultural research institute are used on this study. The five classes are defined as the followings: 1. fine texture, 2. medium and fine texture, 3. medium and coarse texture, 4. coarse texture. 5. shallow soil layer. However, the correlation is not significant between the nitrogen content NH_4^+-N or NO_3-N content in groundwater and classes of soil profile texture in this investigation (Fig. 8,9). However, the results reveal that the potential of the NO_3-N leaching will increase in sandy soil or

shallow soil layer field. Besides, one of the reasons might well be attributed to the heavy rate of fertilizer used by local farmers.

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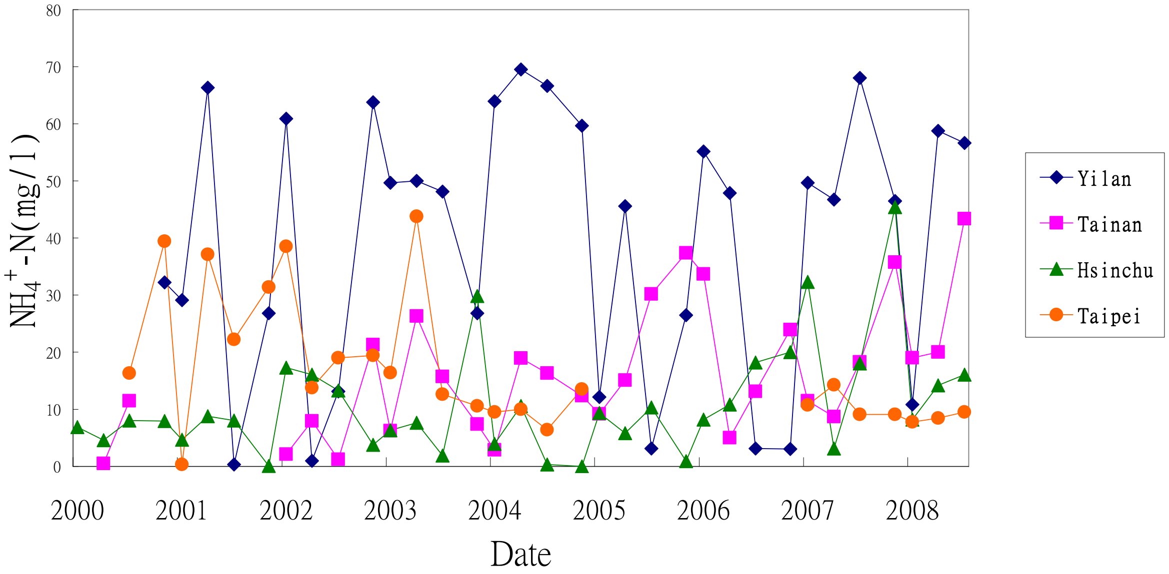

Fig. 1. Higher NH_4^+-N contents in groundwater monitored from some sites by EPA.

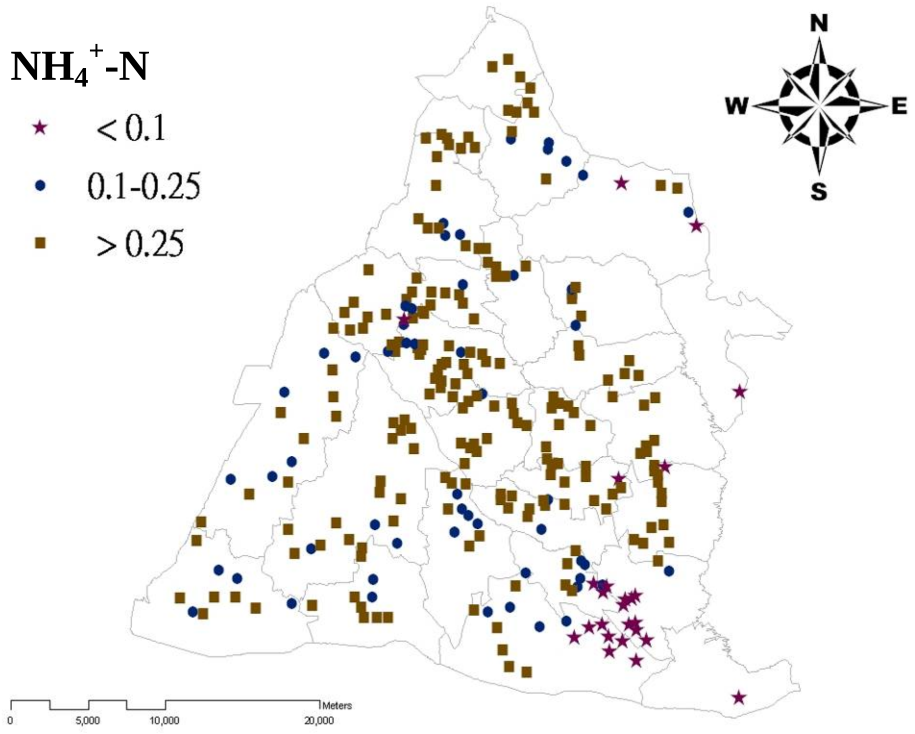

Fig. 2. NH_4^+-N Contents in groundwater monitored in Changhua area.

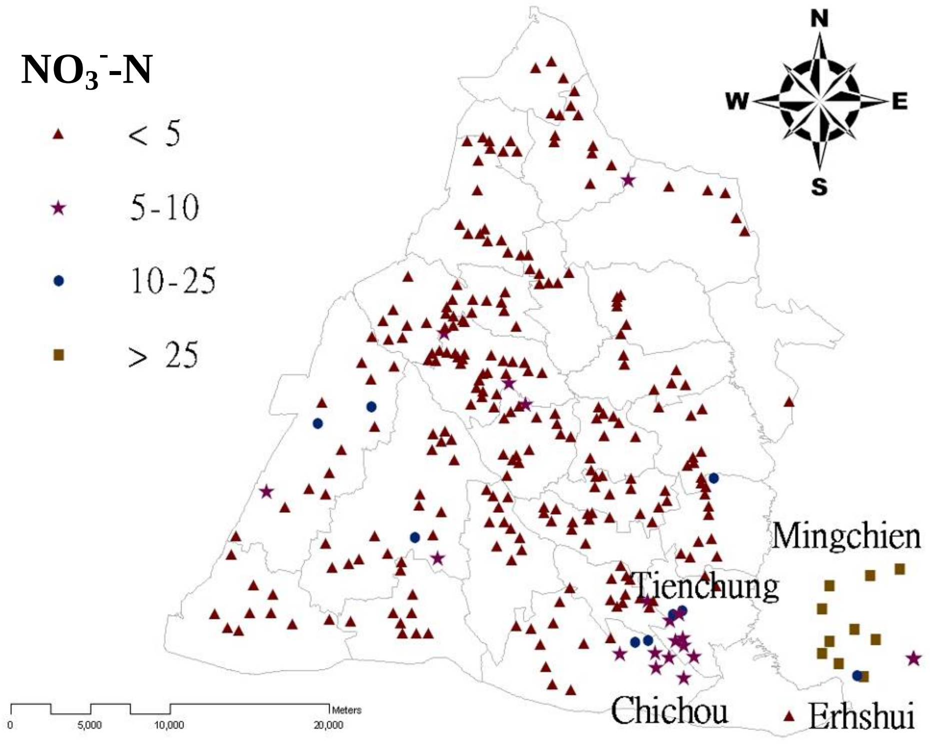

Fig. 3. NO_3^--N Contents in groundwater monitored in Changhua and Nantou areas.

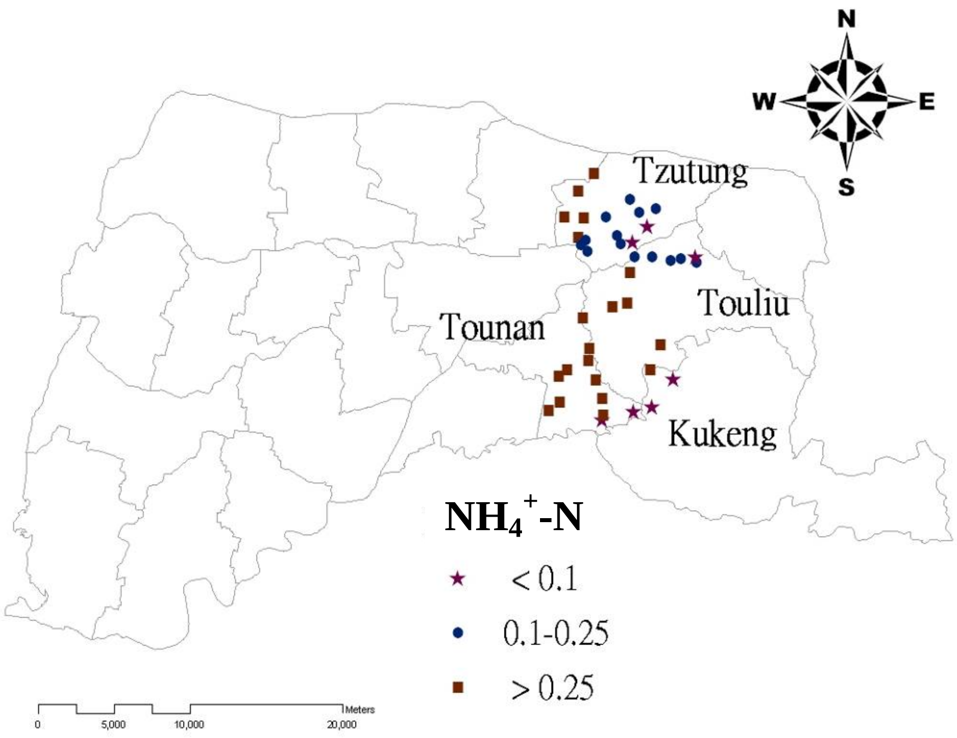

Fig. 4. NH_4^+-N Contents in groundwater monitored in Yunlin area.

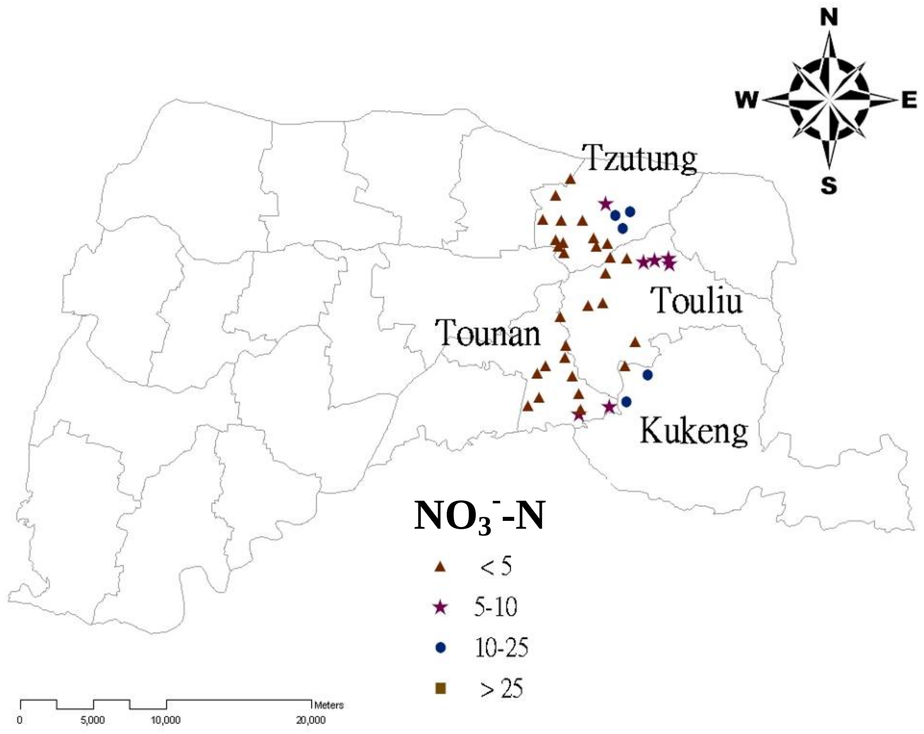

Fig. 5. NO_3^--N Contents in groundwater monitored in Yunlin area.

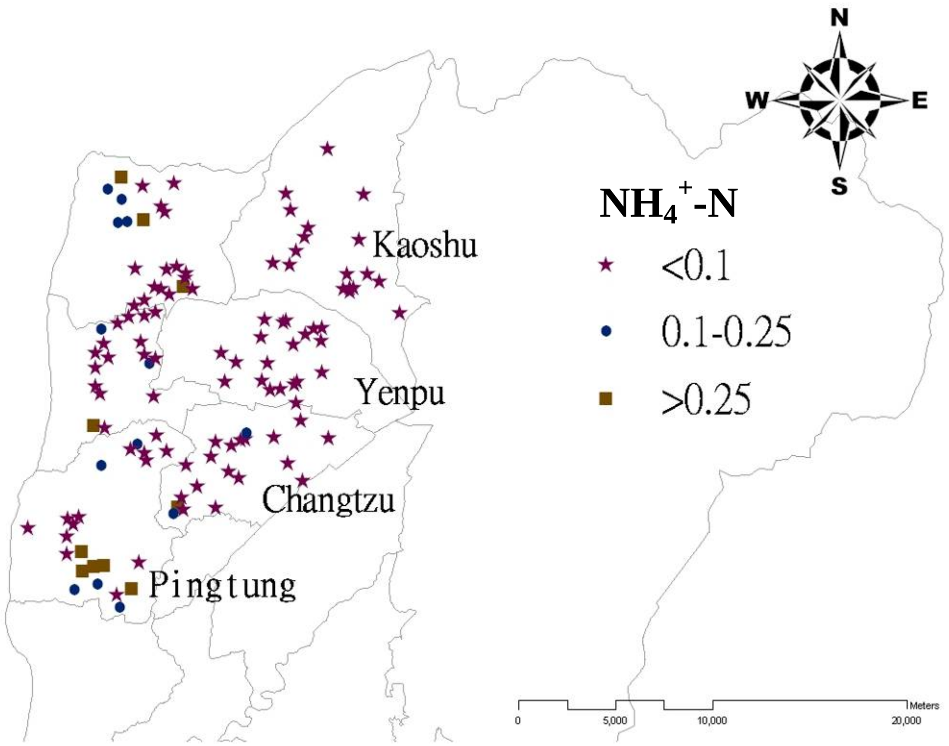

Fig.6. NH_4^+-N Contents in groundwater monitored in Pingtung area.

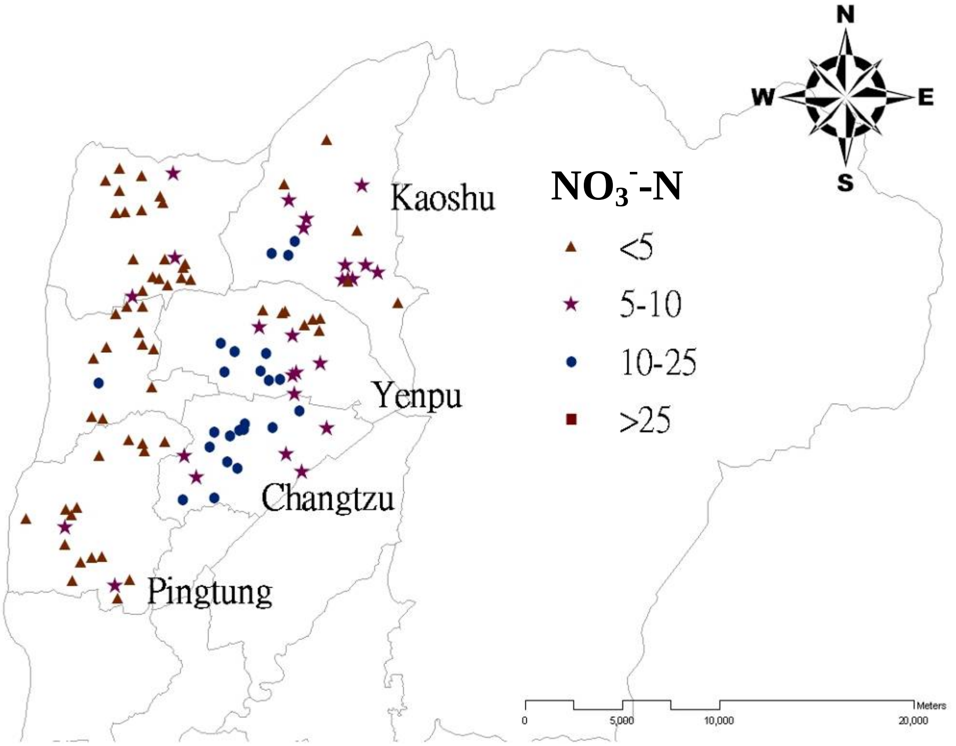

Fig. 7. NO_3^--N Contents in groundwater monitored in Pingtung area.

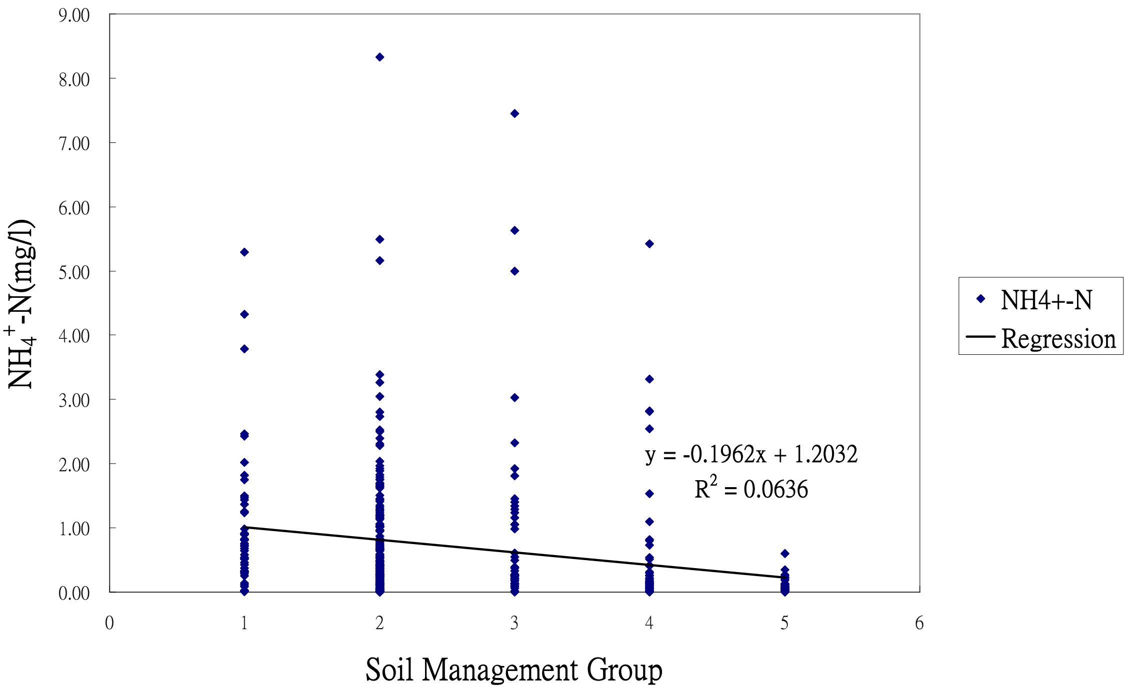

Fig. 8. Relationship between NH_4^+-N contents in groundwater and Soil texture of the total sampled data from some agricultural area.

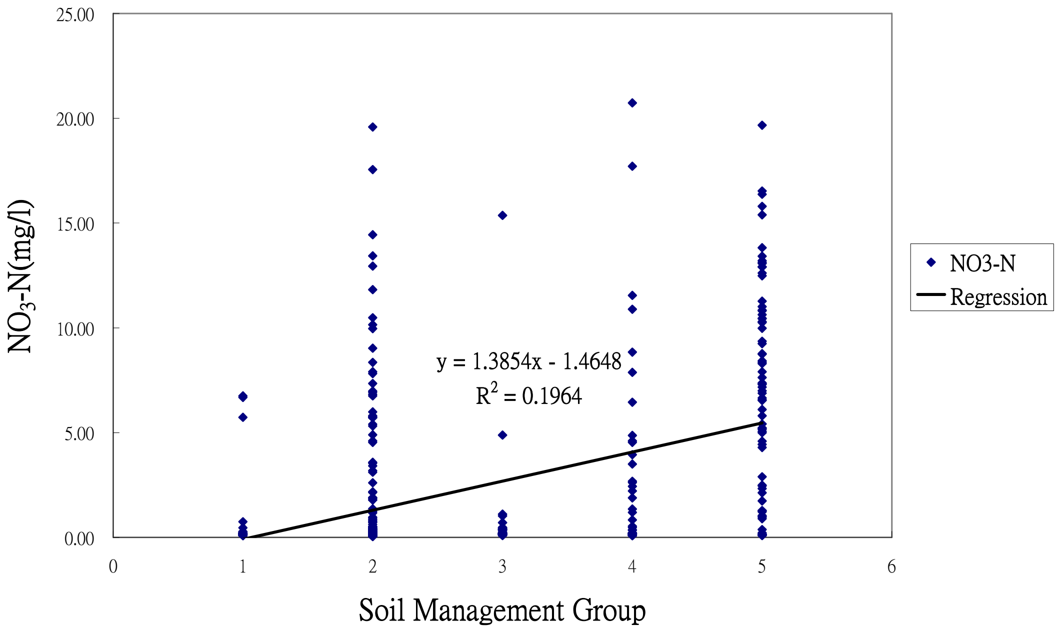

Fig. 9. Relationship between NO_3^--N contents in groundwater and Soil texture of the total sampled data from some agricultural area.

Table 1. Comparison between two networks for monitoring groundwater

Organization	WRA	EPA
Initial year	1958	1995
Purpose	1. Realize the the hydrology of geology 2. Monitor the water resource and quality 3. Monitor the subsidence	1. Monitor the water quality 2. Prevent the contamination
No. of monitoring	244 (575)	394
Depth of well	25 - 400 m	10 - 20 m
Sampling frequency	Once half a year	Once each season
Item	28, including NH_4^+-N, NO_2^--N, NO_3^--N	23, including NH_4^+-N, NO_2^--N, NO_3^--N
Monitoring region	groundwater resource region	1. groundwater resource region 2. Subsidence area
Site of monitoring	School or public organization	School or public organization

Table 2. Groundwater quality standards of various countries

Country	Groundwater Standards (mg/L)				
	Ammonia-N		Nitrate-N		P
N.J.A.C., UK[1]	0.5		10		
Utah, US	0.5		10		
Virginia, US	0.025		5		
North Carolina, US			10		
Wisconsin, US			10		
The Netherland			10		0.15
Taiwan	Type I [2]	Type II[2]	Type I	TypII	
	0.05	0.25	5	25	

1. Groundwater of special ecological significance by National Joint Advisory Council (N.J.A.C.), UK.
2. Type I: groundwater under water sources and quality protection area for drinking water, Type II: Others.

Table 3. Percentages of wells with NH_4^+-N contents in groundwater monitored by WRA higher than water quality limit.

Year	No. of wells monitored	NH_4^+-N (mg/l)	No. of wells with NH_4^+-N > 0.1 mg/l (%)	No. of wells with NH_4^+-N > 0.25 mg/l (%)
2001	368	0.06-16.60	66 (18)	57 (15)
2002	506	0.07-60	139 (27)	131 (26)
2003	547	0.01-39.9	273 (50)	175 (32)
2004	374	0.05-171	240 (64)	225 (60)
2007	450	0.03-259.2	357 (79)	276 (61)

* There are many missing data in 2005, 2006 in the database.

Table 4. Percentages of wells with NO_3^--N contents in groundwater monitored by WRA higher than water quality limit.

Year	No. of wells monitored	NO_3^--N (mg/l)	No. of wells that NO_3^--N > 10 mg/l (%)	No. of wells that NO_3^--N > 25 mg/l (%)
2001	368	0.05-17.8	3 (1)	0 (0)
2002	506	0.04-25.8	1 (0)	1 (0)
2003	547	0.02-48	34 (6)	7 (1)
2004	374	0.06-17.30	2 (1)	0 (0)
2007	450	0.03-3120	6 (1)	3 (1)

* There are many missing data in 2005, 2006 in the database.

Table 5. Percentages of wells with NH_4^+-N contents in groundwater monitored by EPA higher than water quality limit

Year	No. of wells monitored	NH_4^+-N (mg/l)	No. of wells that NH_4^+-N > 0.1 mg/l (%)	No. of wells that NH_4^+-N > 0.25 mg/l (%)
2000	281	0.01-39.4	251(90)	193(69)
2003	413	0.02-50	329(80)	235(57)
2006	395	0.02-55.2	236(60)	165(42)
2007	430	0.02-68	232(54)	168(39)

Table 6. Percentages of wells with NO_3^--N contents in groundwater monitored by EPA higher than water quality limit.

Year	No. of wells monitored	NO_3^--N (mg/l)	No. of wells that NO_3^--N > 10 mg/l (%)	No. of wells that NO_3^--N > 25 mg/l (%)
2000	281	0.01-166	10(4)	2(1)
2003	413	0.02-97.70	38(9)	9(2)
2006	395	0.01-65.40	36(9)	3(1)
2007	430	0.01-41.30	34(8)	3(1)

Table 7. The range of items measured for groundwater

Item	Unit	Range
pH		6.2-8.0
EC	μS/cm	197-3070
SO_4^{2-}	μg/ml	8-913
Cl	μg/ml	3-415
NH_4^+-N	μg/ml	nd-8.3
NO_3-N	μg/ml	0.05-20.7
P	μg/ml	nd-6.3

Table 8. Percentages of wells with NH_4^+-N contents in groundwater of agricultural area higher than water quality limit.

Area	No. of wells monitored	No. of wells that NH_4^+-N > 0.1 mg/l (%)	No. of wells that NH_4^+-N > 0.25 mg/l (%)
Chunghwa	278	253 (91)	199 (72)
Yunlin	41	34 (83)	20 (49)
Pingtung	121	23 (19)	10 (8)
Total	440	310 (70)	229 (52)

Table 9. Percentages of wells with NO_3^--N content in groundwater of agricultural area higher than water quality limit.

Area	No of wells monitored	No of wells that NO_3^--N > 10 mg/l (%)	No of wells that NO_3^--N > 25 mg/l (%)
Chunghwa	278	8 (3)	0 (0)
Yunlin	41	5 (12)	0 (0)
Pingtung	121	25 (21)	0 (0)
Total	440	36 (8)	0 (0)

A Preliminary Proposal for Groundwater Source Protection Zone of the Choshuichi Groundwater Basin

Chung-Jung Chiang[1,2] and Jui-Er Chen[1]

Abstract

Surface water source protection zones have been widely delimited and declared in Taiwan according to the Tap Water Act and Drinking Water Management Statutes. Although the laws are also valid for groundwater regimes, no groundwater source protection area has been set up due to limited information concerning the groundwater systems and concealed groundwater recharge mechanisms. Reviewing study results related to recharge areas, recharge sources, yearly recharge volume and water quality of aquifers, the major recharge areas can be clearly delineated and proposed as the groundwater source protection zone of the Choshuichi Groundwater Basin.

Key word: Groundwater source protection zone

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濁水溪沖積扇地下水水源保護區劃定之芻議

江崇榮 [1,2] 陳瑞娥 [1]

摘要

台灣地區依據自來水法和飲用水管理條例，已針對地表水廣泛劃定水源保護區；此二項法規雖亦適用於地下水水源保護區之劃定，然而因為地下水系統較難掌握，地下水水源保護區劃定工作尚未展開。本文以濁水溪沖積扇地下水區為例，整合過去數年來水文地質及地下水補注相關調查研究成果，評估主要補注區之分布範圍，建議劃定為地下水水源保護區，希望能經由地下水水源之保護，達到地下水資源永續利用之目標。

關鍵詞：地下水水源保護區

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前言

臺灣地區民國78年至87年之年平均用水總量約181.3億立方公尺，其中由河川引水供應75.9億立方公尺（41.9 %），地下水抽用量62.8億立方公尺（34.6 %），水庫供水42.6億立方公尺（23.5 %）；由此可知，地下水供水量為水庫供水量之1.5倍（經濟部水利署，2003）。有鑑於地下水資源之高度重要性，而地下水補注區為地下水之水源地，因此行政院環境保護署民國87年提出之國家環境保護計畫，明訂應調查研擬地下水天然補注區範圍，建立補注區水質保護及污染源管制措施。然而地下水系統埋藏於地下，在缺乏水文地質基本資料及長期地下水水質和水位觀測資料狀況下，地下水天然補注區範圍之劃定及保育工作尚未落實。

經濟部自80年至97年間執行臺灣地區地下水觀測網整體計畫，全程三期共17年，投入大量人力和物力，建置觀測井網，蒐集水文地質及地下水水位和水質基本資料；其中以第一期七年完成建置之濁水溪沖積扇和屏東平原地下水區，資料最為完備；依據相關研究成果，江崇榮和陳瑞娥 (2004) 曾提出屏東平原地下水水源保護區劃定之芻議。本文則針對濁水溪沖積扇，以水文地質、地下水流場及地下水水質特性圈繪補注區範圍；用氫氧穩定同位素質量平衡分析，評估地下水不同補注水源所佔之比率；利用地下水位歷線及地下水層蓄水係數，進行地下水收支分析；據以釐定主要地下水補注區之範圍，建議劃設為地下水水源保護區，希望對保護區土地利用及水土污染做適當管制，使地下水資源管理能朝向永續利用的目標跨出一步。

水源保護區相關法規及劃定概況

水源保護區相關法規主要有自來水法和飲用水管理條例；其與保護區劃定之條文及劃定情形概要如次：

水質水量保護區

自來水事業對其水源之保護，得視實際需要，依水利法向水利主管機關申請辦理，或請水利主管機關會商有關機關，劃定及公布水質水量保護區域，禁止在區域內一切貽害水質與水量之行為（自來水法第十一條）。

自來水公司依據自來水法，以其取水口以上集水區及下游400公尺河川行水區為範圍，並以天然稜線為界線，申辦劃定水質水量保護區；從民國64年起陸續劃定111處水質水量保護區，其中包括水庫水源保護區16處，河川取水口水源水質水量保護區95處，

總面積約9,902平方公里（駱尚廉，2001）；迄民國91年1月，公告廢止四區及縮減高屏溪流域水源水質水量保護區範圍，總面積成爲9,061平方公里，約佔臺灣面積之25 %。地下水雖然也是自來水公司之重要水源，然而並未針對其取水井之地下水水源加以界定及申辦劃定保護區。

水質保護區

行政院環境保護署依民國86年修訂之飲用水管理條例第五條，劃定飲用水水源水質保護區77處，及飲用水取水口之一定距離範圍49處，總面積約3,577平方公里，約佔臺灣面積之10 %（駱尚廉，2001）。其所稱之飲用水水源爲自來水及水質符合飲用水水源之地面水體和地下水體（飲用水管理條例第三條及第六條）。飲用水水源水質保護區及取水口一定距離範圍內，原有建築物及土地使用，經主管機關會商有關機關，認爲有污染水源水質者，得通知所有權人限期拆除、改善或改變使用；其所受之損失由自來水事業或相關事業補償之（飲用水管理條例第五條）。

以上兩法劃定水源保護區之目的，均在保護飲用水之水源；自來水法之水質水量保護區之劃定，是由自來水事業單位提出申請才辦理，主管機關處於被動；而飲用水管理條例之水質保護區，則由主管機關主動評估劃定，因此尚可包括自來水事業以外之飲用水源，劃定之適用範圍較大；惟水質保護區劃定工作起步較晚，而且目前以符合飲用水水源水質且爲甲類水體者爲主要劃定對象，因此已劃定之範圍反而較小，並且大部分與水質水量保護區重疊。地下水雖亦爲臺灣地區之重要飲用水水源，然而目前尚未針對地下水體劃定水源水質保護區；因此佔水資源供應量三分之一的地下水，仍然處於沒有受到積極保育的狀態下。

濁水溪沖積扇地下水水源保護區之劃定步驟

本文所稱地下水水源指的是地下水補注區；地下水補注區可從水文地質結構、地下水流場和水質特徵等釐定其範圍；地下水補注水源，可利用氫氧穩定同位素之質量平衡分析，評估各項補注水源所佔之比率（江崇榮等，2005）；地下水補注量，可利用地下水位歷線分析方法評估（江崇榮等，2006）。經由上述研究工作，可充分掌握地下水補注區、補注源和補注量，以及其在地下水區空間上之分布，據以劃分出主要補注區，以爲地下水水源保護區優先劃定之對象。

（一）以水文地質及地下水流場辨識補注區

經濟部中央地質調查所之濁水溪沖積扇水文地質調查研究總報告（1999）所描繪本區之範圍較大，北以烏溪爲界，向東延伸到車籠埔斷層，向南則擴大到北港溪略南，往西則延展到地下水層推測尖滅之處，據以描述廣域之水文地質及地下水流概念模型；圖1顯示中部地下水層（F2）邊界、地下水流場及剖面aa′、bb′、cc′ 之分布，地下水由東側台地、丘陵及河床區向西側沿海區滲流，烏溪、八卦山脈、濁水溪及斗六丘陵均爲地下水流源頭；圖2、3、4顯示本區300公尺深度內之水文地質，由四層地下水層間夾三層阻水層所組成，阻水層向東尖滅，扇頂附近各地下水層互相連通；補注區由北而南有八卦山脈、濁水溪河床及沖積扇頂區、斗六丘陵等。圖5繪出上部阻水層（T1）由西向東分布之界線T1，以及各地下水層（F1、F2、F3、F4）礫石層由東向西分布之界線，顯示沖積扇頂附近是以巨厚之礫石層爲主所組成，約在T1線以東之扇頂區各地下水層上下互相連通，因此扇頂區之降雨、灌溉水、河水之入滲形成本區各地下水層之共同以及主要補注。

本文以沖積扇內，亦即平原區爲探討範圍（圖6），不包括區外之八卦山脈和斗六丘陵；並以資料較多之上、中、下部（F1、F2、F3）三層地下水層爲對象；而各地下水層之共同補注區位於沖積扇頂區，相當於圖5分佈於T1以東之乙級補注區。

（二）以地下水氚、溶氧和硝酸態氮輔助辨識補注區

氚爲一放射性同位素，半衰期12.43年，天然氚由宇宙線之中子撞擊大氣層中之氮原子而成，並與氧結合成水分子；大氣之氚呈一穩定的濃度值，臺灣地區大氣及降雨含氚之背景濃度約3.0 TU；由於1952年到1962年間進行之大氣層熱核爆試驗，使大氣及雨水氚濃度從1953年起開始急速上升，並於1963年到達最高約600.0 TU（Yu *et al.*, 1998）。假設1952年前入滲補注之地下水背景值3.0 TU至5.0 TU，於1992到1997年間取樣分析，因爲經40多年的衰退，氚濃度約0.3 TU到0.7 TU，而氚分析的精確度（precision）約0.1 TU，若以1.0 TU爲臨界值，當水樣含氚大於1.0 TU，可認定其應爲1953年或更晚入滲補注之水，或至少混有1953年後補注之水，代表該水樣取樣於補注區或其臨近。水中之溶氧主要來自大氣或水生植物。未受污染之地面水，在25 ℃左右之溶氧濃度約8 mg/l（Mc Neely *et al.*, 1979）。當地面水入滲補注地下水後，往深處或下游滲流，因阻水層封閉及微生物及化學作用而漸被耗盡；因此地下水高溶氧之分布區可顯示出地下水補注之位置。以0.5 mg/l 爲臨界值，大於0.5 mg/l 表示顯著含溶氧補注區之指標，小於此值則爲含氧極微之非補注區。

雨水中之硝酸態氮（NO_3^-–N）一般在0.2 mg/l以下，未受污染之地面水常在1.0 mg/l以下

（Mc Neely *et al*., 1979）。由於氮肥及人畜排泄物污染土壤和地面水，可使補注區地下水之硝酸態氮濃度遠高於1.0 mg/l ；地下水往深處或下游滲流，因阻水層封閉及微生物作用下，硝酸態氮與水中溶氧幾乎同步被還原而迅速耗盡，基於上述特性，硝酸態氮可爲地下水污染及補注區之指標，高濃度之區域爲被污染區，也是地下水快速補注區（江崇榮等，2005）。

氚、溶氧和硝酸態氮爲示踪劑，三者均由入滲水帶入地下水層中，可指示地下水補注區之分布；先以氚大於1.0 TU爲條件，再以溶氧或硝酸態氮大於0.5mg/l 爲附加條件來輔助劃定補注區範圍，其界線如上部地下水層和中部地下水層之R1和R2線（圖6）；上部阻水層（T1）爲上部（F1）和中部（F2）地下水層之分隔層，T1線以東爲主要補注區之初估範圍，面積約273平方公里；至於R1和R2，則以氚、溶氧和硝酸態氮標示出的補注區範圍；兩種方法完全不同，然而所獲界線卻相當一致，可互相驗證其合理性。由於上部地下水層被覆於中部地下水層之上，地下水區內之垂向補注必先進入上部地下水層，再向下補注中部地下水層，受兩層之間阻水層之封隔，中部地下水層之補注區理應小於上部地下水層之補注區。圖六卻顯示界線R2在R1的西側，此乃中部地下水層是濁水溪沖積扇之主要抽水層次，地下水水力坡度和流速較大，示踪劑傳輸速度較快距離較遠所致。綜上所述，本文採用R1界線以東爲上部地下水層和中部地下水層之共同補注區，也是濁水溪沖積扇之主要地下水補注區，面積約285平方公里（江崇榮等，2005），佔全區面積2079平方公里之13.7 %。

（三）以氧穩定同位素組成評估補注水源

在水文循環中，水中之氧穩定同位素組成，只因蒸發或凝結時之同位素分化作用，以及不同水源之混入而有所改變；同一地點之地下水、地面水或雨水之氫氧穩定同位素組成值常維持在一定範圍內；藉由地下水及補注水源氫氧穩定同位素之質量平衡分析，可估計出地下水是由補注源按多少的比率混合成的。圖7和表1摘錄本區氧穩定同位素質量平衡分析之結果（江崇榮等，2005），顯示濁水溪沖積扇地下水區之主要地下水補注區內，降雨入滲補注約佔總補注量之22.9 %；上游河水流入扇頂區於河床入滲補注約佔總補注量之34.0 %，兩項合計約56.9 %；亦即在面積僅佔全區13.7 % 的主要補注區內，補注量達全區之56.9 %。因此可知濁水溪的水質和水量，以及主要補注區土地使用型態和污染程度，是控制濁水溪沖積扇地下水補注水質和水量之主要因子。

（四）地下水補注量評估

彰化雲林地區因缺乏水庫，民生用水、工業用水和農業用水均高度仰賴抽取地下水供應；然而因抽水井常未裝置水表，地下水抽用量難以明確統計，導致地下水收支平衡分析具有高度不確定性。比較直接而可靠的方法是地下水歷線分析方法（江崇榮等，2004，2005）；將日平均地下水位乘上蓄水係數及代表面積，加總後可得代表全區在該日之蓄水量，據以繪製全年之365日之地下水（量）歷線；每年約10月起至次年1月爲乾季，濁水溪沖積扇地下水歷線之退水線呈穩定之直線 (L)，其斜率可代表全年之穩定抽水率；2月至3月之乾季地下水歷線之退水線，則呈另一較大斜率之直線 (M)，其斜率代表穩定抽水率加上水稻灌溉抽水率；利用上述L、M直線之斜率可估算出全年之穩定抽水量和一、二期作水稻灌溉之抽水量，加總得年抽水量 (Qp)。此外，以10月至次年1月之乾季退水線 (L) 之平行線，將各次退水線往後延伸到因降雨而上升歷線之頂點下，延伸線與歷線頂點之垂直差距代表各場次降雨之補注量，加總可得全年補注量 (Q_r)。歷線在年底與年初之垂直差距，則代表全年蓄水變化量 (Q_s)。最後，應用水平衡式，$Q_r = Q_P + Q_\ell + Q_s$，可計算出地下水之流失量 (Q_ℓ)。地下水歷線分析法評估結果如表2，88、89及90年之年抽水量依次約13.65、14.58、11.73億立方公尺，補注量依次約15.36、15.34和15.30億立方公尺，蓄水變化量依次約-1.33、-0.94和1.17億立方公尺，水平衡公式計算得流失量依次爲3.04、1.70和2.40億立方公尺（江崇榮等，2006）。3年之補注量甚接近，平均值爲15.33億立方公尺，若乘上氧同位素質量平衡分析所獲之補注水源之補注率，可概估河水在濁水溪沖積扇頂河床年入滲補注約5.21億立方公尺（圖8），降雨在扇頂區入滲補注約3.51億立方公尺；亦即主要補注區佔全區總面積13.7 %，入滲補注總水量達8.72億立方公尺。

（五）地下水污染潛勢之考量

從圖2、3、4概念模型及水文地質剖面得知，只有沖積扇頂是主要非受壓區，補注水於此入滲及分到各層之中，是地面水入滲補注入口之主要途徑，也是污染物傳輸之門戶。圖9上部及中部地下水層中，81年到89年間地下水取樣分析硝酸態氮濃度分佈概況（陳文福等，2003），來自地表之硝酸態氮污染，集中在沖積扇頂；由此可看出兩件事，一是沖積扇頂爲地下水易受污染區，另一是地下水已遭受相當程度的人爲污染。硝酸態氮污染主要來自氮肥及人畜排泄物，是一種廣泛的區域性污染；地下水中硝酸態氮的濃度大於1 mg/l，顯示極接近污染源，可爲補注區之另一指標。沖積扇頂是上、中、下部地下水層之共同補注區，區內土壤或地面

水污染極易傳輸到深淺各地下水層中，是全地下水區地下水水質和水量保育的重點區域。

（六）地下水水源保護區劃定之建議

本研究釐定之地下水主要補注區面積約285平方公里，佔濁水溪沖積扇地下水區之13.7 %；年補注水量約8.72億立方公尺，佔全區之56.9 %；面積小補注量大，而且位居地下水流之源頭，土地或水體若受污染或不當之利用，將嚴重影響全區地下水之水質和水量；建議劃設為地下水水源保護區。正式劃定保護區時應由主管機關再確認其範圍，進行必要水文地質補充調查，以提高保護區界線之精確度。

地下水水源保護區建議管制事項

自來水法第十一條，訂定11項禁止貽害水質與水量之行為。飲用水管理條例第五條禁止12項污染水源水質之行為，可為地下水水源保護區管制之基礎；經歸納並參考德國地下水保護區管制內容（江崇榮，1985；DVGW，1975），建議管制事項如次：

一、以營利為目的之飼養家畜家禽。

二、有機或無機肥料、農藥或除草劑之過量使用。

三、排放超過規定標準之工、礦廢水或家庭污水。

四、掩埋或棄置垃圾、灰渣、污泥、糞尿、廢油、廢化學品、動物屍體或其它足以污土壤、地表水或地下水之物品。

五、非法砍伐林木、開墾土地或採取土石。

六、河道變更或河川工程足以影響河床滲透或河水自淨能力且未經主管機關同意者。

七、新社區開發或新墳場之設立。

八、工業區開發，污染性工廠設立或污染物品輸送管線之設置。

九、道路、鐵路、捷運系統、機場、軍事設施等之開發，未經主管機關同意或未經環境影響評估通過者。

十、核子反應器之設置或運作，放射性廢料之儲存或處置。

水源保護區屬於公益性限制發展區，對於公益提供者，依其損失或付出之大小，應給予適當補償，其經費來源可比照自來水法第十二條或飲用水管理條例第五條之規定，由自來水事業提供，亦可考慮徵收地下水水權費或抽水費來支應。水質水量保護區依都市計畫程序劃定為水源特定區者，其土地應視限制程度減免土地增值稅、贈與稅及遺產稅（水利法第十二

條之一)。

地下水水源保護區上游集水區之水，亦爲地下水之補注水源；以濁水溪爲例，約有34.0 %之地下水源自於濁水溪上游集水區之河水，34.2 %源自東側緊接的八卦台地和斗六丘陵區；此等區域因在地下水區外，未被本文納入保護區一併探討，惟其重要性不容忽視，尤其台地及丘陵地茶園和果園發達，其使用之農藥和肥料也可能間接流入地下水區中，值得在未來進一步調查探討。

結論與建議

地下水是臺灣地區之重要飲用水源，保護此項資源以達永續利用之目標，乃刻不容緩的工作。截至目前爲止，重要地表水體已依據自來水法劃定自來水水源水質水量保護區，飲用水管理條例劃定飲用水水源水質保護區；此二項法令雖然涵蓋地下水體，然而因對地下水之了解有限而遲未劃定其水源保護區。近年來臺灣地區地下水觀測網計畫建立起量多質佳之水文地質及地下水資料，對地下水系統有較確實之掌握，提供地下水水源保護區劃設工作之科學依據。居於地下水在水資源供應上佔有高比重、不可替代性及易受污染，主要補注區之保育工作實爲迫切的重大課題。

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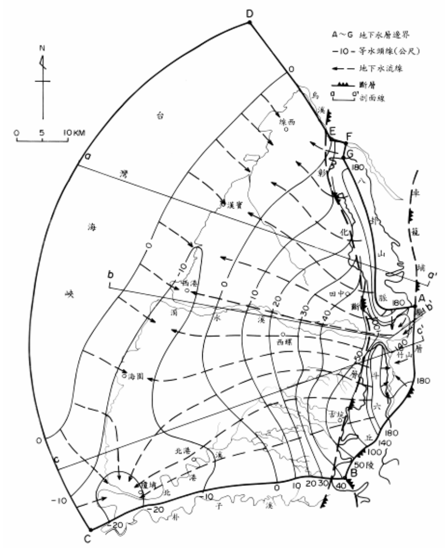

圖 1 . 濁水溪沖積扇中部水層（F2）之邊界及地下水流概念模型（圖中數值係 1997 年 8 月和 1998 年 4 月平均水位）。

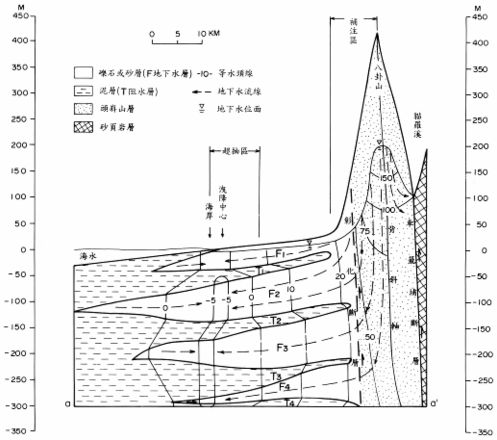

圖 2. 濁水溪沖積扇 aa′ 剖面水文地質及地下水流概念模型（圖中數值係 1997 年 8 月和 1998 年 4 月平均水位）。

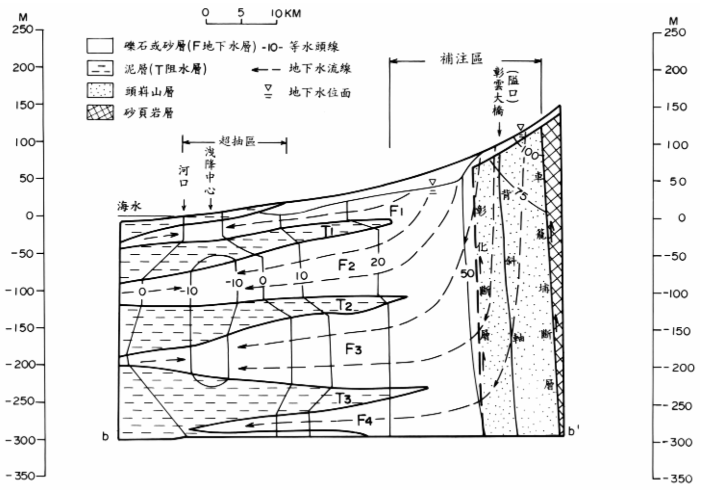

圖 3. 濁水溪沖積扇 bb′剖面水文地質及地下水流概念模型（圖中數值係 1997 年 8 月和 1998 年 4 月平均水位）。

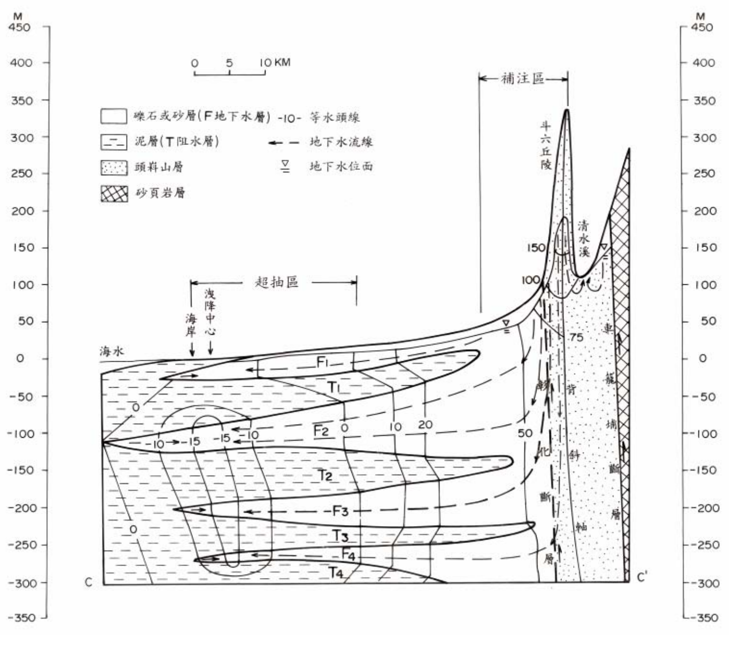

圖 4. 濁水溪沖積扇 cc′ 剖面水文地質及地下水流概念模型（圖中數值係 1997 年 8 月和 1998 年 4 月平均水位）。

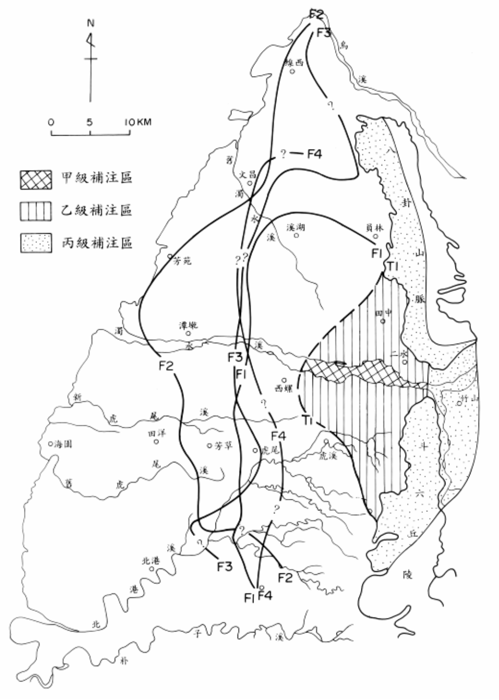

圖 5. 濁水溪沖積扇地下水補注區及上部阻水層界線 T1 與各地下水層（F1、F2、F3、F4）之礫石層由東向西分布界線。

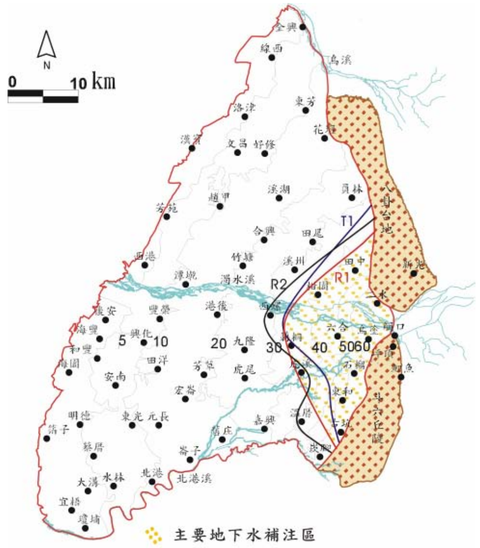

圖 6. 氚、溶氧和硝酸態氮輔助辨識之濁水溪沖積扇上部及中部主要地下水補注區分布略圖。

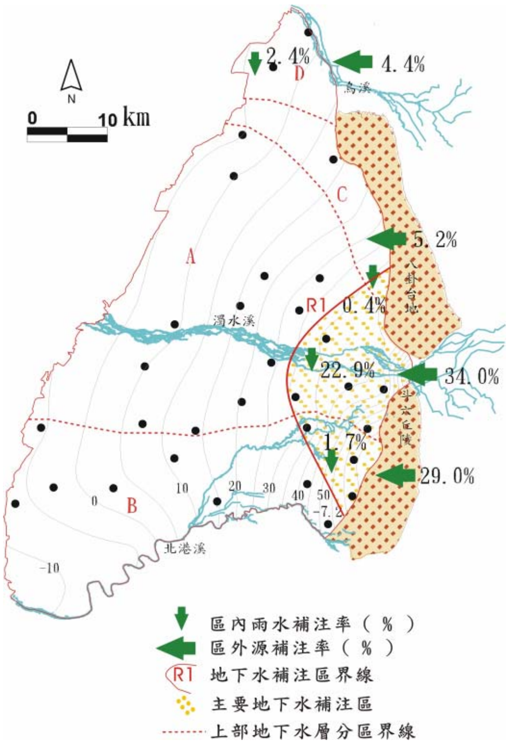

圖 7. 濁水溪沖積扇主要地下水補注區及地下水補注源分布圖。

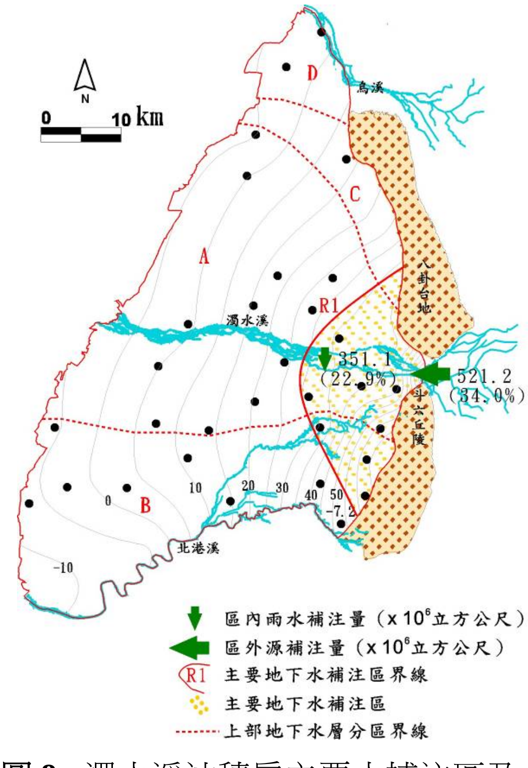

圖 8. 濁水溪沖積扇主要水補注區及 88 至 90 年平均地下水補注量分布圖。

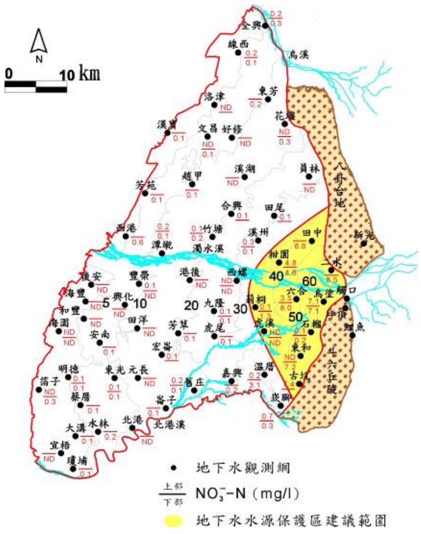

圖 9. 濁水溪沖積扇上部（F1）及中部（F2）地下水層歷年 NO3--N 濃度分佈概況所顯示之污染潛勢。

表 1. 各分區內源和外源在全區所佔之比率

A 濁水溪分區	濁水溪水 X ＝ 0.598 × 207.216 ÷ 364.084 ＝ 0.340 ＝ 34.0 %
	雨水（1－X）＝ 0.402 × 207.216 ÷ 364.084 ＝ 0.229 ＝ 22.9 %
B 北港溪分區	斗六丘陵地下水 X ＝ 0.945 × 111.826 ÷ 364.084 ＝ 0.290 ＝ 29.0 %
	雨水（1－X）＝ 0.055 × 111.826 ÷ 364.084 ＝ 0.017 ＝ 1.7 %
C 員林花壇分區	八卦台地地下水 X ＝ 0.925 × 20.564 ÷ 364.084 ＝ 0.052 ＝ 5.2 %
	雨水（1－X）＝ 0.075 × 20.564 ÷ 364.084 ＝ 0.004 ＝ 0.4 %
D 烏溪分區	烏溪水 X ＝ 0.649 × 24.487 ÷ 364.084 ＝ 0.044 ＝ 4.4 %
	雨水（1－X）＝ 0.351 × 24.487 ÷ 364.084 ＝ 0.024 ＝ 2.4 %
外源總比率 ΣX ＝ 0.726 ＝ 72.6 % 內源總比率 Σ（1－X）＝ 0.274 ＝ 27.4 %	

表 2. 濁水溪沖積扇地下水收支估計值($Q_r = Q_P + Q_\ell + Q_s$，$\times 10^6$ 立方公尺)

年	補注量 Q_r	抽水量 Q_P	流失量 Q_ℓ	蓄水變化量 Q_s	淨補注量 $Q_r - Q_\ell$
88	1,536	1,365	304	-133	1,232
89	1,534	1,458	170	-94	1,364
90	1,530	1,173	240	117	1,290
平均	1,533	1,332	238	-37	1,295

Nutrient Interception by a Riparian Forest Buffer Strip from Adjacent Orchard near Reservoir: Two Case Studies in Taiwan

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Summary

Two case studies, including Techi reservoir located at Taichung county, central Taiwan, and Feitsui reservoir located at Taipei county, northern Taiwan, were selected to study the effect of nutrient interception by a riparian forest buffer strip receiving the adjacent orchard region by applying the high and low quantity of chemical fertilizers, respectively. The widths of riparian forest buffer strip are at least 50 meters in these two case study sites. The composition, spatial variability and seasonal changes of cations and anions in soil solution collecting from orchard, interface and riparian buffer zones were analyzed to control the water quality in two water reservoirs. The results indicated that the soil nutrients in the orchard near Techi reservoir, especially for the ions of phosphate, K, Ca, Mg, and ammonium-N, can be significantly intercepted by a forest buffer strip within 30 meters. Owing to no any application of chemical fertilizers in the orchard near Feitsui reservoir since 1987, the nutrient concentration of different ions in soil solution is very low. This low concentration of ions in soil solution released from orchard region near Feitsui reservoir can be intercepted in the interface area between the orchard region and forest buffer strip. The results also indicate that the width of riparian buffer strip needs at least 10 meters to intercept the soluble nutrients under this very low input of chemical fertilizers. We make sure that enough width, at least 30 meters of riparian buffer strips for these two case studies, can be regarded as a nutrient sink to intercept the soil nutrients soluble from orchard region and keep good water quality in the reservoirs.

Key Words: Riparian forest buffer strip, buffer width, eutrophication, soil solution, water quality, interception, nutrient sink.

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1. Introduction

The forest has a function of conserving the source of water, reducing the soil erosion, reducing the earth's surface runoff and blocking the nutrients to move. The riparian forest buffer strip of the forestry deals in and collects in the water area is very important for keeping of river quality. Hewlett (1985) pointed out that applying a large amount fertilizer above and below of the forest buffer strip, if there is forest buffer strip, then the concentration of nutrients of the groundwater does not have obvious change, but under having forest buffer strip, the concentration of nutrients in the groundwater will rise by a wide margin, and will influence the quality of the stream. It also indicated that forest buffer strip take the importance to safely guard the quality of water source in the forest.

The riparian buffer strip means that it is the transition ecosystem area between ecosystem of land and water ecosystem, this area is often the important nutrient source of a water ecosystem. Therefore, riparian buffer strip is the important factor considered to maintain the productivity of river basin. The riparian buffer strip means there is circumference of a river or reservoir which has a width of the plants of forest, bush or grasses, etc. It has certain interception or filters the function of the nutrients flowed from both sides of river, and it does not cause the eutrophication for reservoir or rivers.

A lot of research papers have pointed out that the riparian buffer strip can buffer the nutrients depend on the difference of the pollutant by buffering width, there must be 10 to 60 meters at least, it can play its function and can protect the water quality of the river or reservoir (Ding and Chen, 1981, 1984; Hsieh and Chen, 1989; Chen *et al*., 1989; Lowrance, 1992; Jordan, 1994; Hsieh *et al*., 1994; Chen and Liu, 1994, 1995; Chen, 1995). In addition, Iowa State University Agroforestry Research Team (IStART) (1994) also indicated that the more plant varieties planting at the both sides of river, for instance trees, short bush and grass as the riparian buffer strip, the more promotion the water quality after the four years buffer function.

Lowrance (1992) indicated that the function of denitrification will be very fast if the groundwater table of the riparian forest soil is within 60 centimeters from surface soil. Therefore, it can remove a large number of nitrate nitrogen from the groundwater in the forest. Jordan et al. (1993) also indicated that riparian forest buffer strip can significantly intercept the nitrogen flow from the corn field which located at both sides of corn field, in term of nitrate nitrogen, organic carbon and soluble organic nitrogen. The strong reduction condition with high electric potential (Eh) (>500 mv) can increase the denitrification function. In addition, Arthur and Fahey (1992 and 1993)

also indicated that seasonal changes of groundwater do not significantly influence the soil chemical properties. Ambus and Lowrance (1991) proposed that denitrification function is to change the nitrate nitrogen of soil solution to produce different influences because of different situations located at the different areas.

Lowrance *et al.* (1984a) and Pinkowski *et al.* (1985) indicated that riparian forest buffer strip can regard as nutrient sink and can reduce the nitrate nitrogen, total nitrogen, calcium, and magnesium which flowed from the area of agricultural cultivation of both sides of river. In addition, research also indicated that the net retention percentage to nutrients of forest buffer strip are N 68 %, Ca 39 %, P 32 %, Mg 23 %, Cl 7 % and K 6 %, respectively (Lowrance *et al.*, 1984a, 1984b; Chichester, 1976; Kilmeret *et al.*, 1974).

Hsia *et al.* (1994) indicated that there are no obvious differences of phosphorus and potassium concentrations of the soil solution in the artificial forest zone of *cryptomeria* between applying fertilizer and without applying fertilizer. The ammonium nitrogen in the soil solution of bush forest zone is all very low, the nitrate nitrogen concentrations is relative high in the surface water only, the other nutrients are all not different compared to those of applying fertilizer. In addition, this study also indicated that the buffer function of the nutrient in the soil of the riparian zone, are dependent on the soil factor, land use, growth condition of cover plant, and the ability of absorbing nutrient of plant roots. In addition, many researches also indicated that the riparian forest zone can absorb the majority of nitrogen and phosphorus flowed from the farmland, orchard or tea garden, and it can intercept about 70-90 % of total nitrogen and nitrate-N in the width of 20 meters of riparian zone (Lowrance *et al.*, 1984c; Peterjohn and Correll, 1984; Jacobs and Gilliam, 1985; Haycock and Pinay, 1993).

King *et al.* (1984) indicated that the heavy rain can produce the soil suspension particle flowed into the stream after the riparian forest was clear cut. All the nutrients concentrations are higher than those of regulation of nutrient ions in the river, including magnesium, potassium, nitrate-N and sulfuric acid. It is obvious that we can make a good soil and water conservation practice to protect the water quality in the natural forest ecosystem. Romkens *et al.* (1973, 1974) also indicated that using a large amount of fertilizer and applying the urine of livestock for increasing the highest crop productivity will promote a large amount of phosphorus concentration flowing into the river system through surface runoff. Therefore, it will accelerate the eutrophication of rivers, creeks and water reservoir. Taiwan Province Government (1981) proposed a technical method to set up the riparian buffer strip in the both sides of reservoir which the width of riparian buffer zones should be at least

15 to 30 meters in the horizontal distance.

Lowrance *et al*. (1988) have studied and pointed out, in the good area of planting and growing, Cs-137 brings in the forest highest from bounded content in place of 30 meters, later reduced sharply. Copper *et al*. (1983) and Copper and Gillian (1987) showed that the riparian forest strip can deposit Cs-137 and enough width of riparian forest buffer strip can block the movements of the nutrients effectively. In addition, Jordan *et al*. (1993) also use bromine ion as the trail of pharmaceutical, and then concluded that 30 meter of forests buffer width can intercept the soil nutrients. Lin (1988) indicated that, at least 10 meters of buffer width, can effectively reduce the concentrations of phosphoric acid and total phosphorus in the soil solution. Chen (1995), Chen and Liu (1994, 1995) and Chen *et al*. (1996) found that salt content of various nutrition in the soil solution is quite low in trial zone of about 10 meters between orchard and forest Feitsui reservoir.

The objective of this article is to review the function of riparian buffer strips, also known as riparian forest zone, on intercepting the excess nutrients discharged from the upper part of the fertilizers or pesticides application area in two case studies in Taiwan (Ding and Chen, 1981, 1984; King *et al*., 1984; Lin, 1988; Chen, 1989, 1995; Chen and Liu, 1994, 1996; Hsia *et al*., 1994; Hsieh *et al*., 1989, 1995).

2. Materials and Methods

2.1 Techi reservoir near Taichung city, Central Taiwan

The first study site was located at Techi reservoir in central Taiwan near Taichung, which the elevation is about 1400 meter (Fig. 1). The soil sampling regions are shown in the left side. The orchard regions are region A to region I. The interface regions are region X, Y and Z. The riparian forest buffer zones are region J, K, L, M, N, P and Q region. The control regions are S and T. The soil solution sampling regions are shown in the ride side. The orchard regions are region A and B. The interface regions are region C and D. The riparian forest buffer zones are region E, F, G, H, I, J, K and L regions. The control regions are Y and Z. The study project was conducted from 1988 to 1989.

The precipitation of the site is 2600 mm/yr and mean annual air temperature is 17 ℃. The site of study soil is classified as Dystrochrepts, an Inceptisols with ochric epipedon in the upper 20 cm and low base saturation less than 70 %. 50 tons/ha of organic fertilizer and mixed chemical fertilizer including 100 kg ammonium sulfate/ha and 200 kg/ha of calcium perphosphate, and 100 kg/ha of potassium chloride were applied into the study site from March to April every year. Mixed chemical

fertilizer including 200 kg potassium chloride/ha and 200 kg/ha of calcium perphosphate is also applied into the site on August every year.

2.2 Feitsui reservoir near Taipei city, Northern Taiwan

The second case study site was located at Feitsui reservoir in northern Taiwan near Taipei, which the elevation is about 300 meter (Fig. 2). The rain water sampling regions are shown with number from 1 to 5. The orchard regions are region 1 and 2. The interface regions are region 3. The riparian forest buffer zones are region 4 and 5. The control regions are region 6. The soil solution sampling regions are shown with English letter from A to J. The orchard regions are region A to region F. The interface region is region G only. The riparian forest buffer zones are region H and I regions. The control regions are J. The study project was conducted from 1993 to 1995.

The precipitation of the site is 3400 mm/yr and the mean annual air temperature is 27 ℃. The site of study soil is classified as Dystrochrepts, Inceptisols with ochric epipedon and low base saturation less than 70 %. No any chemical fertilizers were supplies since 1987.

2.3 Design of sampling of soil solution in experimental sites

One orchard size of 30 m x 50 m and one riparian forest buffer strip of 60 m x 50 m were selected as experimental plot in each site. Two separate forest areas of 20 m x 20 m were also selected as the forest control site.

Four and 8 collection apparatus of soil solution were set up in the orchard area and forest buffer strip of Techi site, respectively, sampling at the site after every storm from January, 1988 to December, 1989.

Six and 3 collection apparatus of soil solution were set up in the orchard and forest buffer strip of Feitsui site, respectively, sampling at the site after every storm from July, 1993 to June, 1995.

2.4 Chemical analysis of soil and soil solutions

The items of soil physical and chemical properties of two sites were determined including soil pH, organic carbon content, cation exchange capacity (CEC), exchangeable bases including Ca, Mg, K, and Na, extractable ammonium-N and nitrate-N, and Bray's No. 1 extractable P.

The items of cations and anions of soil solutions were determined including pH, electric conductivity, K, Na, Ca, Mg, ammonium-N, nitrate-N, phosphate-P, F, Cl, Br, and Sulfate ions.

3. Results and Discussion

3.1 Variation of soil nutrients in Techi site

The distribution of different soil characteristics and all kinds of nutrition in the different

regions collected from different sampling time during the period from 1987 to 1989 (Fig. 3). The soil pH ranged from 3.8 to 4.9, which increased with increasing soil depth. The results also indicated that nutrients, such as phosphorus, potassium, calcium and magnesium sampled from riparian forest strip are higher than those of orchard region. Nitrate-N and sodium have significantly differences between orchard and riparian forest strip region. The potassium, sodium and magnesium concentration at orchard, riparian forest strip intersection and forest buffer strip all have similar monitoring concentration. In surface soil samples, ammonium-N concentration in riparian forest strip is significantly higher than that of orchard region, it is result in the high soil organic matter content in the riparian forest strip. The various nutrients of the subsurface soil samples in different research districts also have the same trend.

The data analysis and spatial variation of surface soil (0-20 cm depth) and subsurface soils (20-40 cm depth) in 1988 showed that pH value has the trend of decrease, but pH has no significant differences between orchard area to riparian forest buffer strip. Ammonium-N, nitrate-N, extractable phosphorus, potassium, magnesium, organic matter, total nitrogen amount and organic nitrogen content, in riparian forest buffer strip of 1st district that have the tendency to significantly increase, then significantly decrease in the 2nd district and the 3rd district of riparian forest buffer strip.

This result also revealed that if we can do good soil and water conservation practices and there is enough width of reservoir forest regions, then the nutrients in soil of orchard district can only be moved to the 1st district of forest buffer strip, about 10 meters from the orchard district, and the concentration of different nutrient ions have already obviously reduced in the collected soil solution.

3.2 Variation of soil solution nutrients in Techi site

The mean concentration of various ions and variations in different soil solution samples of the research area are shown in Fig. 4. Because the farmers used the chemical fertilizer all the year in the study site before the experiment, potassium, calcium, magnesium, available phosphorus and total phosphorus content in soil solution are all have significantly higher concentration in the orchard district. But, the available phosphorus and total phosphorus content drop rapidly in the intersection region between orchard district and forest buffer strip, and keep a low level content in the forest buffer strip. It revealed that the phosphorus in the soil of this research district can pollute the water quality of groundwater or reservoir. Because ions are moved differently between potassium, sodium, calcium, and magnesium ions, therefore, there are different distribution situations shown in the soil

solution. Chlorine ion and ammonium nitrogen had higher amounts in the upper zone of riparian forest buffer strip, which has significantly differences compared to those of other nutrients. It means that the concentration of Na, Ca, Mg, phosphate and total phosphorus are significantly decreased in the riparian forest strip. The proposed width of riparian forest buffer strip is at least 30meters in Techi site.

3.3 Principal component analysis (PCA) in Techi site

1. The Principal component analysis of soil nutrients

Selecting surface soil samples (0-20 cm) sampling on February 9, 1988, as an example, after analyzing of various nutrients by principal component analysis (Fig. 5). The result of this PCA analysis indicated that the variation of first two principal components can account for 85.6 % of total variance in Techi site. Among them, the first principal component accounts for 62.5 %, and the second principal component accounts for 23.1 %. The first principal component can distinguished obviously soil sample of orchard district and forest buffer strip, and the bounded area between orchard district and forest buffer strip can also defined. Because of applying chemical fertilizers and changes of topography, it makes the relatively great variability of various nutrient contents in the surface soil samples of orchard district. This can also be shown by the analysis result of the first axis of PCA. In addition, potassium, sodium, calcium, magnesium and extractable phosphorus content can reach positive correlation more than 0.86. The subsurface soil samples also have the same trend as surface soil samples by principal component. It revealed that potassium, sodium, calcium, magnesium and extractable phosphorus contents in the soils which have the high to low concentration gradient change from orchard district to forest buffer strip.

2. The Principal component analysis of soil solution nutrients

The principal component analysis of average nutrient contents in soil solution of the Techi site are shown in Fig. 6. It clearly reveal that the samples of the lower forest zone (regions of F, G, H, I, K, L), higher forest zone (regions of C, D, E, J) and forest control area (region Y & Z) were regarded as the same groups and they were distinctly different from the orchard samples (regions A and B). The result of this study also revealed that the variation of the first principal component accounts for 51.5 % of total variation, and the principal component with calcium, magnesium, extractable phosphorus and total phosphorus content have apparent positive correlation. The variation of second principal account for 23.9 % of total variation, and the principal component with ammonium-N and chloride contents has apparent positive correlation. It is noteworthy that the nutrient content of lake water sample was closer similar to that of the samples collected from the

interface regions between orchard area and forest zones (Fig. 6). We also found that the contents of sodium, calcium and magnesium of lake water are relatively higher than those of riparian forest buffer strip.

3.4 Variation of soil nutrients in Feitsui site

This surface soil and subsurface soil samples near ten soil solution sites were analyzed for soil pH value and various nutrients. The pH value ranged from 4.1 to 4.6 except forest soil in check area, which is slightly high. The soil organic carbon content of surface soil and subsurface soil samples are relatively high, ranged from 20 to 35g/kg.

Because this research area has not applied the chemical fertilizer since 1987, therefore, the concentrations of different cations and anions in soil or soil solution are very low. For the nutrients, there is no variation tendency on the space in surface soil (0-20 cm) and subsurface soils (20-40 cm). The result also indicated that the organic nitrogen content account for more than 93 % of the total soil nitrogen content in all samples, except few samples. There is a tendency of decreasing from orchard area to forest buffer zones for available phosphorus content, reducing from 160 mg/kg to 11 mg/kg. The organic phosphorus content in study site has 80-90 % of the total phosphorus content in most surface soils and subsurface soils in the study.

3.5 Variation of soil solution, rainwater and reservoir lake water nutrients in Feitsui site

The pH, EC value and the concentrations of potassium, sodium, calcium and magnesium in soil solution, rainwater and reservoir lake water are shown in Figs. 7, 8, and 9. The pH value does not have obvious change in soil solution from Fig. 7, ranged from 4.9 to 5.1. In forest check area, the pH value is 5.8. The pH value of rainwater is about 5.6. The pH value of reservoir lake water is about 6.7, based on the standard values in reservoir water quality ranged from 6.5 to 8.5. The EC value of soil solution and rainwater is quite stable ranged from 0.04 to 0.07 dS/m, except very few samples. This may have something to do with the plant decomposing or enrich effect. The EC value of reservoir lake water is quite stable, which revealed that the changes with time is difficult at the present stage in water quality of Feitsui site.

In general, the concentrations of different cations and anions in soil solution are very low (Fig. 8 & 9). The ions of K, Na, Ca and Mg in soil solution are always found in slightly higher concentration in interface region between orchard area and forest buffer strip, and these results also indicated that this interface area can be regarded as a nutrient sink shown as Fig. 8 and Fig. 9. The average concentration of sodium ion in reservoir lake water is higher than those of soil solution. In Fig. 9, we also found that the concentration of calcium, magnesium are less than 1.0 and 0.2 mg/L

in the soil solution, and there is no any effect on the water quality of Feitsui reservoir.

The concentrations of ammonium-N, nitrate-N, fluorine ion, chlorine ion, phosphate ion and sulfate ion in soil solution, rainwater and reservoir lake water (Fig. 10 to 12). From Fig. 10, there is no any ammonium-N and phosphate ions can be detected in the soil solution even in the orchard area. Only one can be found in the rainwater sample is its highest concentration in 1.4 mg/L, but the average concentration is under 0.1 mg/L. The concentration of nitrate-N is almost less than 1.0 mg/L, even more in orchard area near detection limit. The concentration of fluorine ion is also quite low, only forest buffer strip and reservoir lake water is slightly high, about 0.3 mg/L, and the others sample in the area was under 0.2 mg/L. There is no significant variation in the study area for these ions. The average concentration of chlorine ion studied in the site is about 5 mg/L and the reservoir lake water is slightly high, about 7 mg/L (Fig. 11).

In addition, the concentration of phosphate ion near zero in different water samples, which revealed that there is no any eutrophication phenomena in the Feitsui reservoir. About sulfate ion, it is the lowest in the rainwater, about 2.5 mg/L. It is slightly higher from orchard area to forest buffer area, about 4 mg/L. There is the highest concentration of sulfate ion in the reservoir lake water, about 8 mg/L, perhaps, which was produced by the soil solution flowing into the reservoir directly or by rainfall. Therefore, the proposed width of riparian forest buffer strip in Feitsui site should be at least 20 meters.

3.6 Spatial and seasonal variation of ions in soil solution of Feitsui site

Because of concentration of various nutrients in soil solution is not high, so that the spatial variations of water quality in different sampling times are also acceptable in the study area. But it still has higher value in the orchard area. In addition, the pH value and the concentration of K and Na ions in the soil solution have a relatively higher spatial variation. The seasonal variations have significantly differences depend on the kind of ions and quantity of precipitation for Ca, Mg, Nitrate-N and phosphate ions in the site. It is very important that the practices of soil and water conservation in hillside need more notice to prevent the eutrophication phenomena was occurred.

4. Conclusion

Nutrients in the soil solutions of orchard near Techi reservoir located in central Taiwan, especially for the ions of phosphate, K, Ca, Mg, and ammonium-N can be significantly intercepted by a forest buffer strip within about 30 meters based on the case studies during 1988 to 1989.

Low concentration of soil solution in the orchard near Feitsui reservoir also can be intercepted

in the interface area between the orchard and forest buffer strip, which also indicated that the width of riparian buffer strip needs at least 10 meters to intercept the soluble nutrients under no any application of chemical fertilizers in the experimental sites during 1993 to 1995.

We can make sure that enough width, at least 30 meters, of riparian buffer strips for these two case studies can be regarded as a nutrient sink to intercept the excess soil nutrients soluble from orchard region and to keep the good water quality in the reservoirs based on these two case studies in Taiwan.

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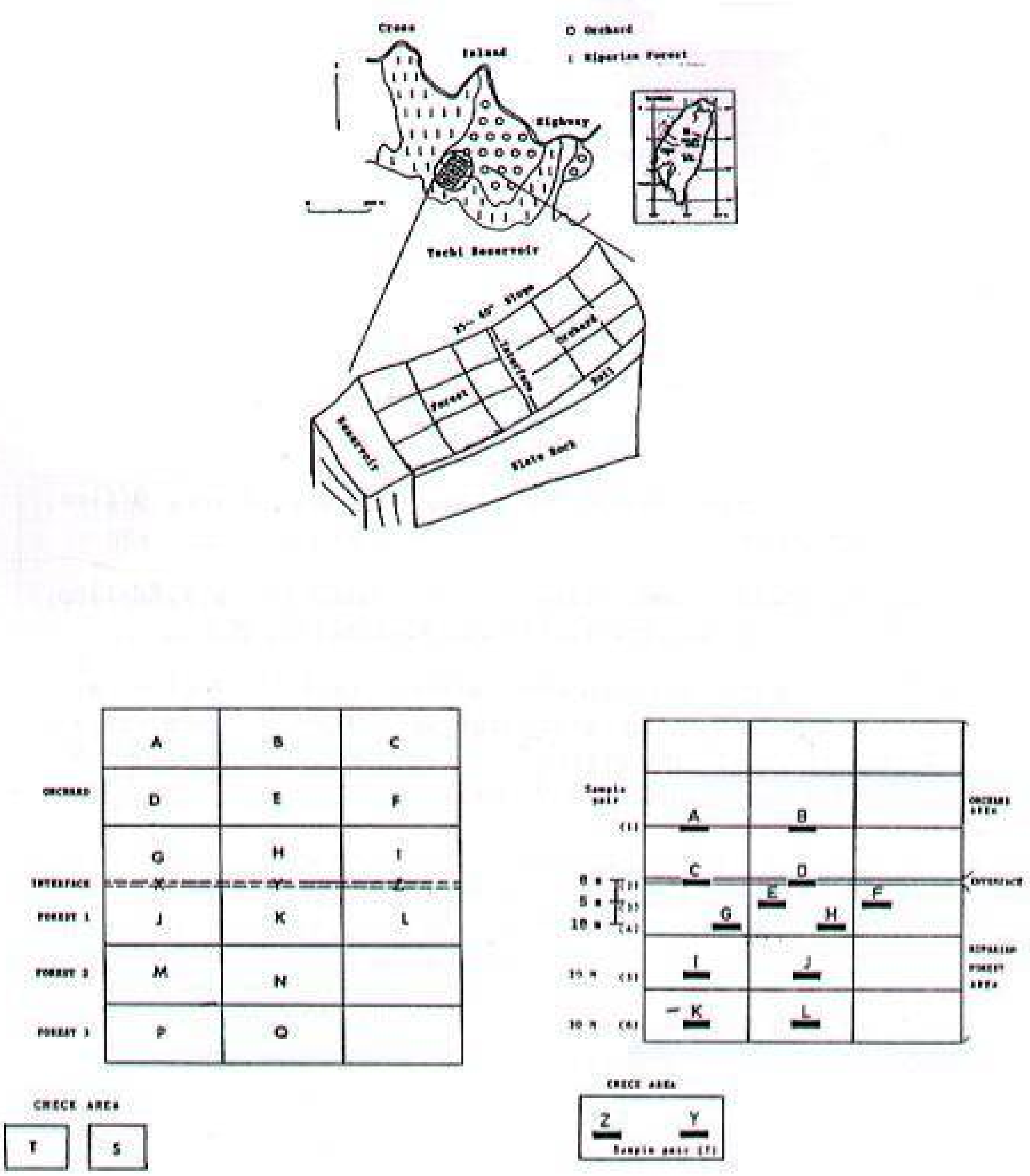


Fig. 1. The geomorphology and sampling sites of Techi reservoir study area in 1988 and 1989.

Notes: The soil sampling regions are shown in the left side. The orchard regions are region A to region I. The interface regions are region X, Y and Z. The riparian forest buffer zones are region J, K, L, M, N, P and Q region. The control regions are S and T.

The soil solution sampling regions are shown in the ride side. The orchard regions are region A and B. The interface regions are region C and D. The riparian forest buffer zones are region E, F, G, H, I, J, K and L regions. The control regions are Y and Z.

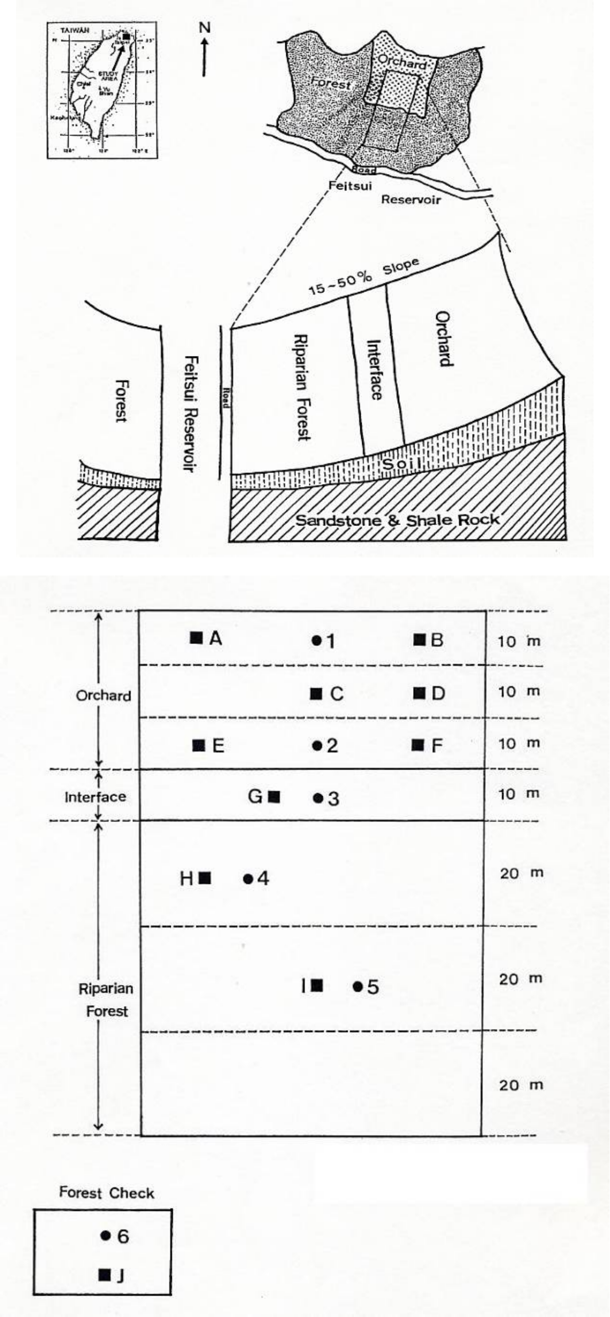


Fig. 2. The geomorphology and sampling sites of Feitsui water reservoir study area in 1993 and 1994.

Notes: The rain water sampling regions are shown with number from region 1 to 5. The orchard regions are region 1 and 2. The interface region is region 3. The riparian forest buffer zones are region 4 and 5. The control region is region 6.

The soil solution sampling regions are shown with English letter from A to J. The orchard regions are region A to region F. The interface region is region G only. The riparian forest buffer zones are region H and I. The control region is J.

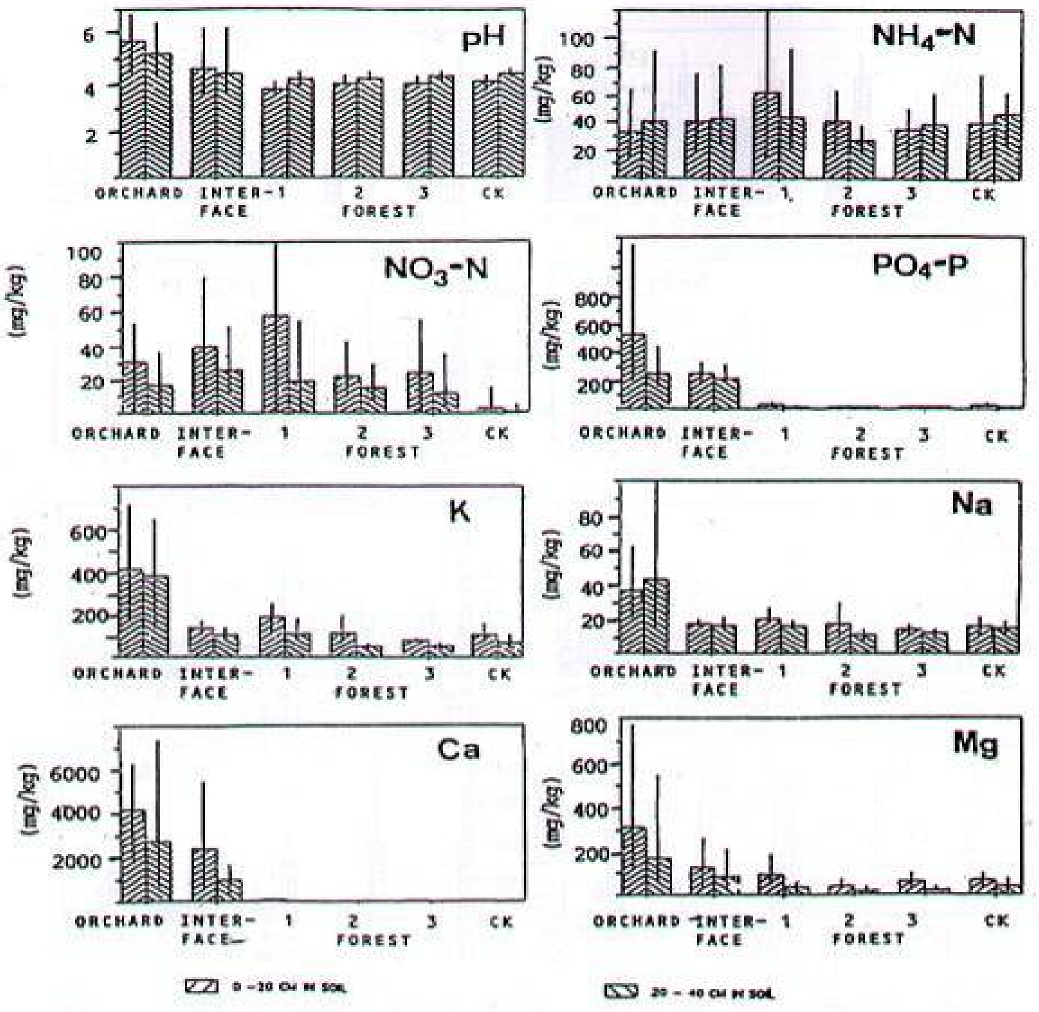
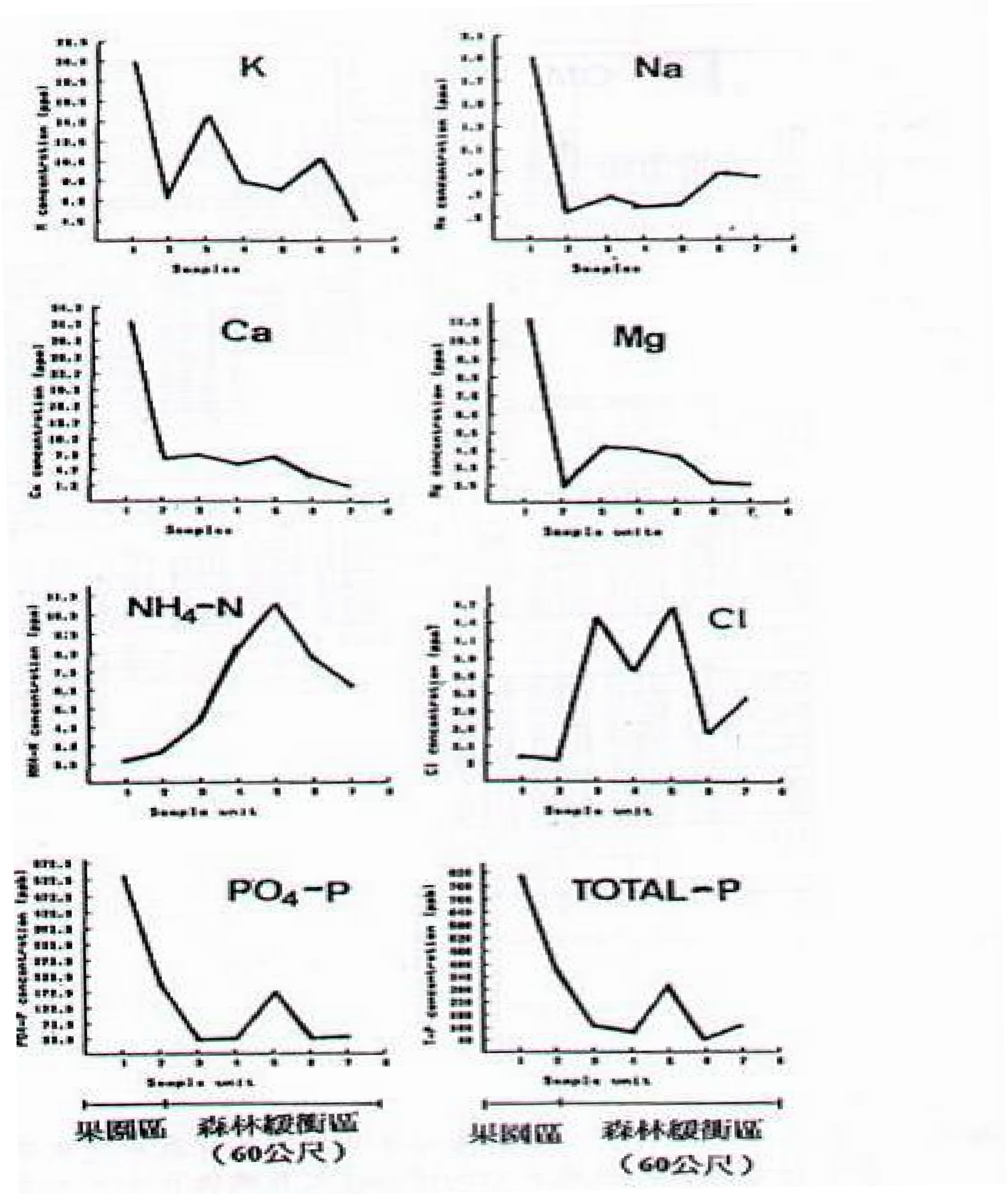


Fig. 3. The distribution of different cations and anion ions in the different soil depths (0-20 cm in the left side and 20-40 cm in the right side for each ions) collected from different sampling area of Techi reservoir.

Notes: Number 1, 2, and 3 are riparian forest zones.



Orchard region   Riparian forest region (60 meters width)   Orchard region   Riparian forest region (60 meters width)

Fig. 4. The distribution of different cations and anion ions in the soil solution collected from different sampling area of Techi reservoir.

Notes: Number 1 is orchard region, #2 is interface region, and from # 3 to #8 are riparian forest zones and the width of this buffer zones is 60 meters.

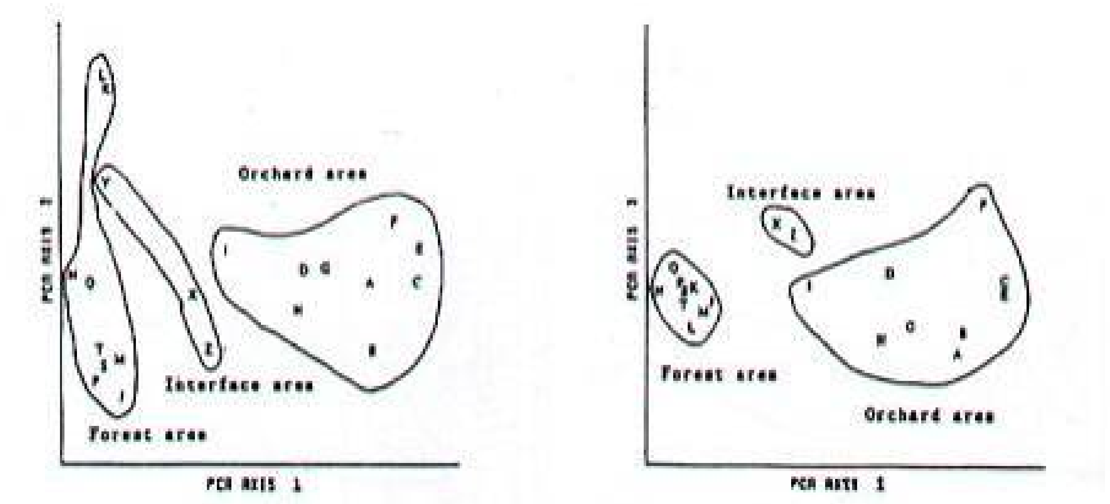


Fig 5. The distribution of the soil samples (0-20 cm depth) collected in 1988, relative to the first three principal component axes of the Techi reservoir.

Notes: The soil sampling regions are shown in the left side of fig. 1. The orchard regions are region A to region I. The interface regions are region X, Y and Z. The riparian forest buffer zones are region J, K, L, M, N, P and Q. The control regions are S and T.

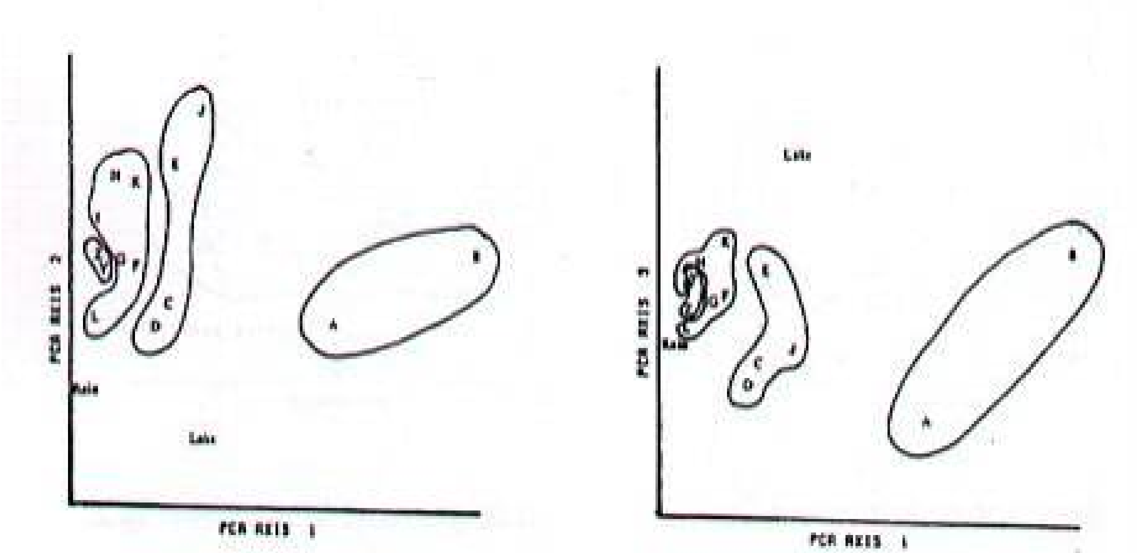


Fig. 6. The distribution of cations and anions of the soil solutions collected in 1988 relative to the first three principal component axes of the Techi reservoir

Notes: The soil solution sampling regions are shown in the ride side of Fig. 1. The orchard regions are region A and B. The interface regions are region C and D. The riparian forest buffer zones are region E, F, G, H, I, J, K and L regions. The control regions are Y and Z.

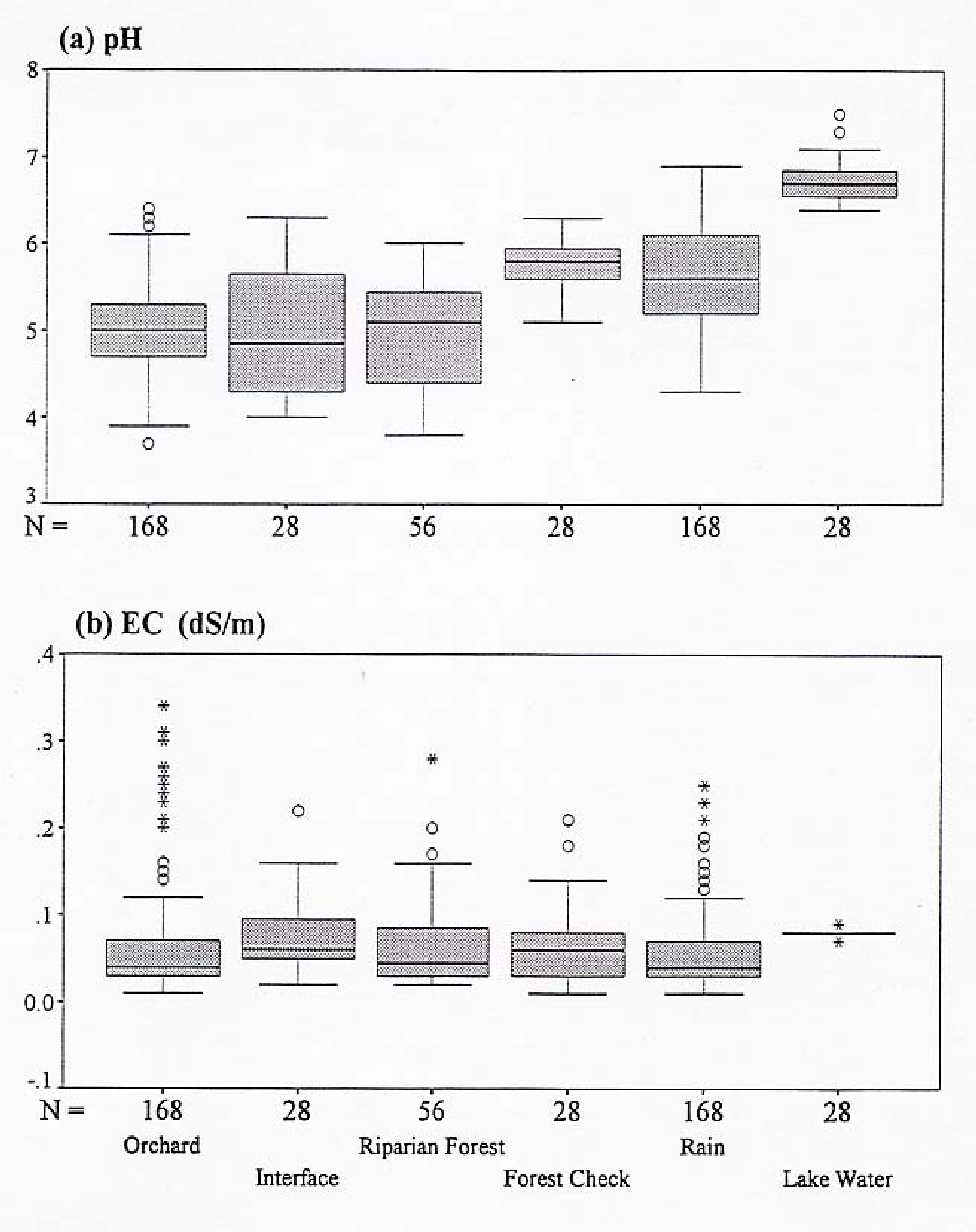


Fig. 7. The distribution of pH values and EC values in the soil solution collected from different sampling area of Feitsui reservoir near Taipei city.

Notes: The values in the black boxes ranged from 25th % to 75th % of total range. The black bar is medium value (50 % of total range). The values of the bars ranged from minimum value to maximum value. The circle value (O) is the 1.5 times of observation values of black boxes far from the 75th % of the total range values. The star value (*) is the 3 times of observation values of black boxes far from the 75th % of the total range values. N is total sampling number.

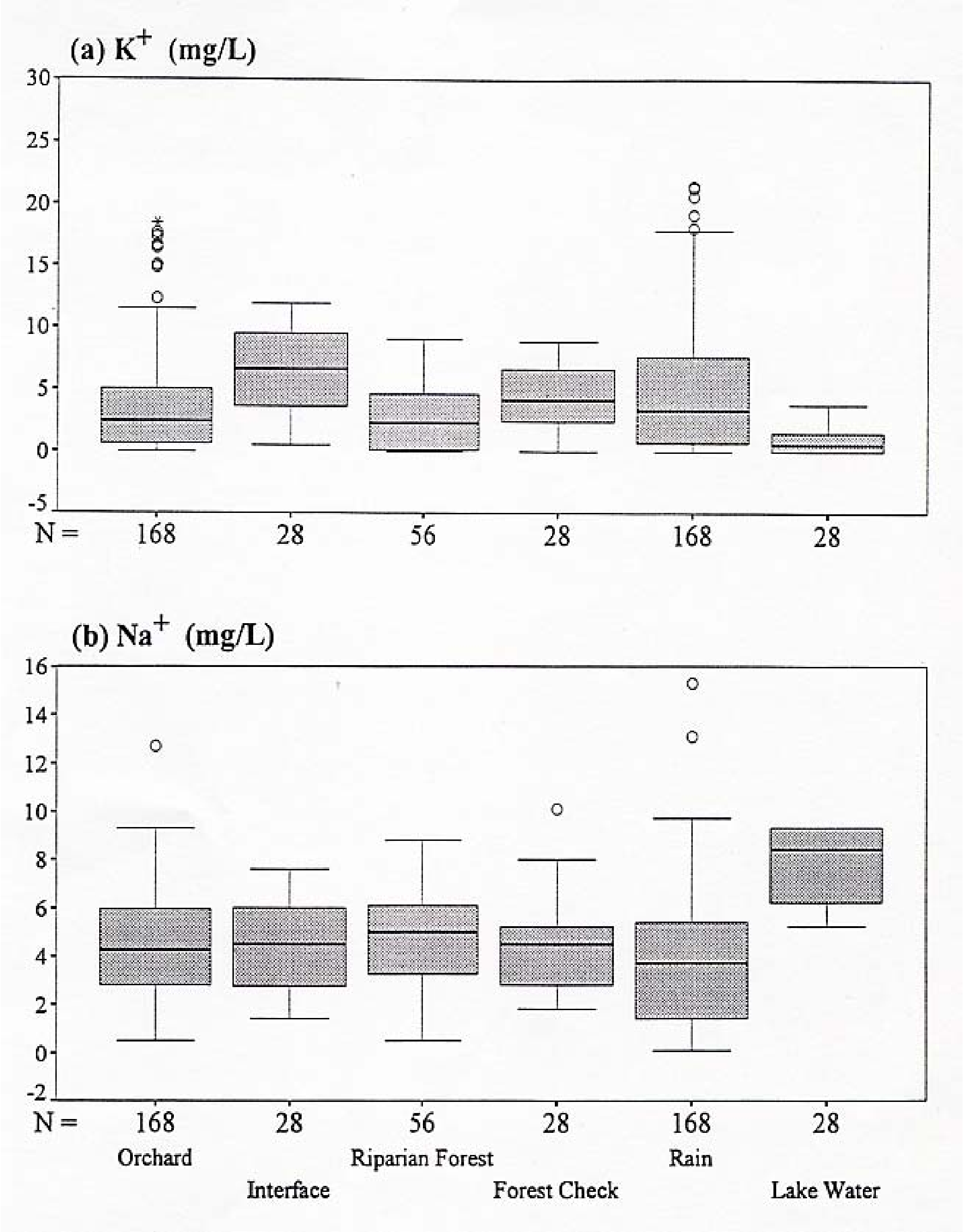


Fig. 8. The distribution of potassium and sodium ions in the soil solution collected from different sampling area of Feitsui reservoir near Taipei city.

Notes: The values in the black boxes ranged from 25th % to 75th % of total range. The black bar is medium value (50 % of total range). The values of the bars ranged from minimum value to maximum value. The circle value (O) is the 1.5 times of observation values of black boxes far from the 75th % of the total range values. The star value (*) is the 3 times of observation values of black boxes far from the 75th % of the total range values. N is total sampling number.

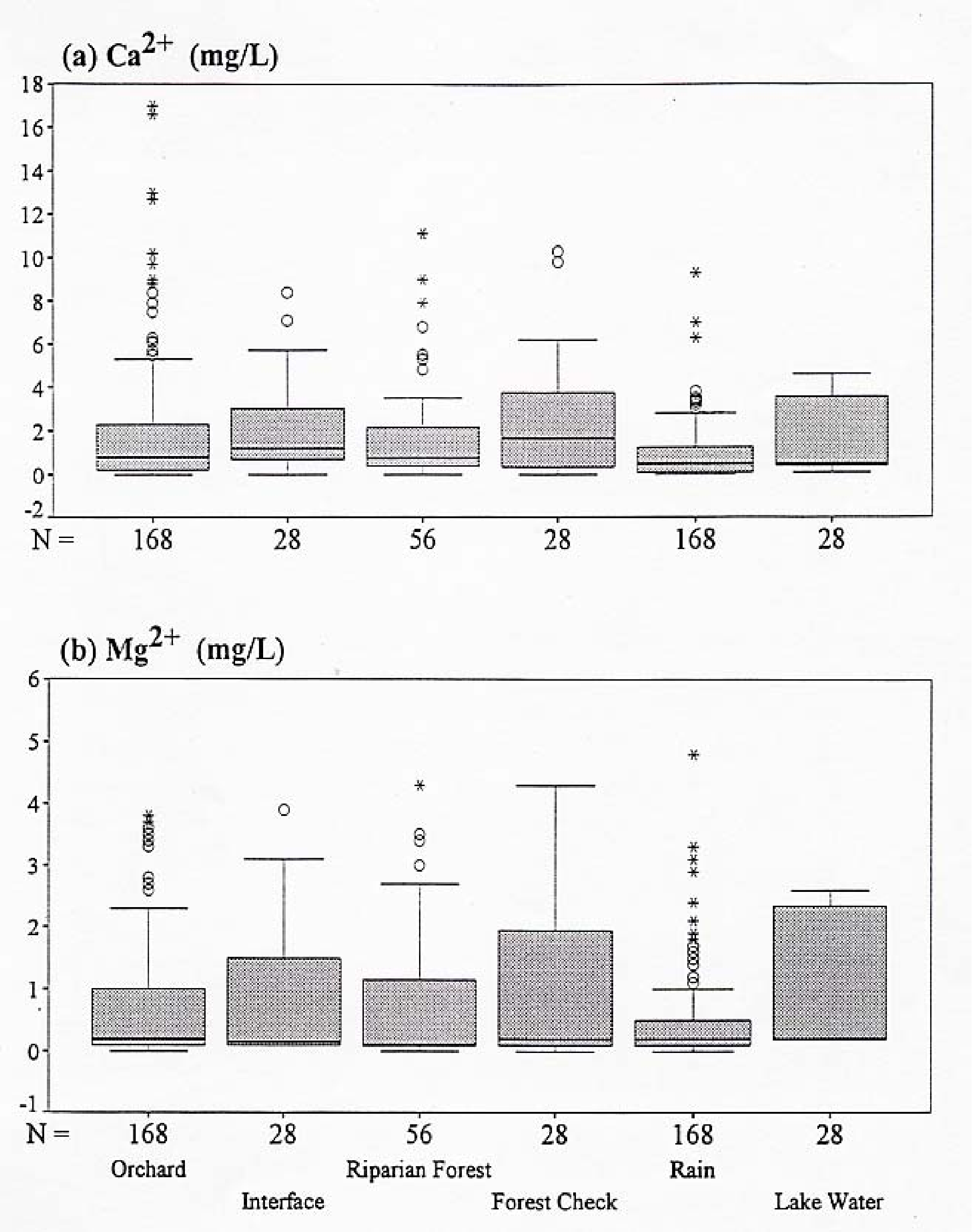


Fig. 9. The distribution of calcium and magnesium ions in the soil solution collected from different sampling area of Feitsui reservoir near Taipei city.

Notes: The values in the black boxes ranged from 25th % to 75th % of total range. The black bar is medium value (50 % of total range). The values of the bars ranged from minimum value to maximum value. The circle value (O) is the 1.5 times of observation values of black boxes far from the 75th % of the total range values. The star value (*) is the 3 times of observation values of black boxes far from the 75th % of the total range values. N is total sampling number.

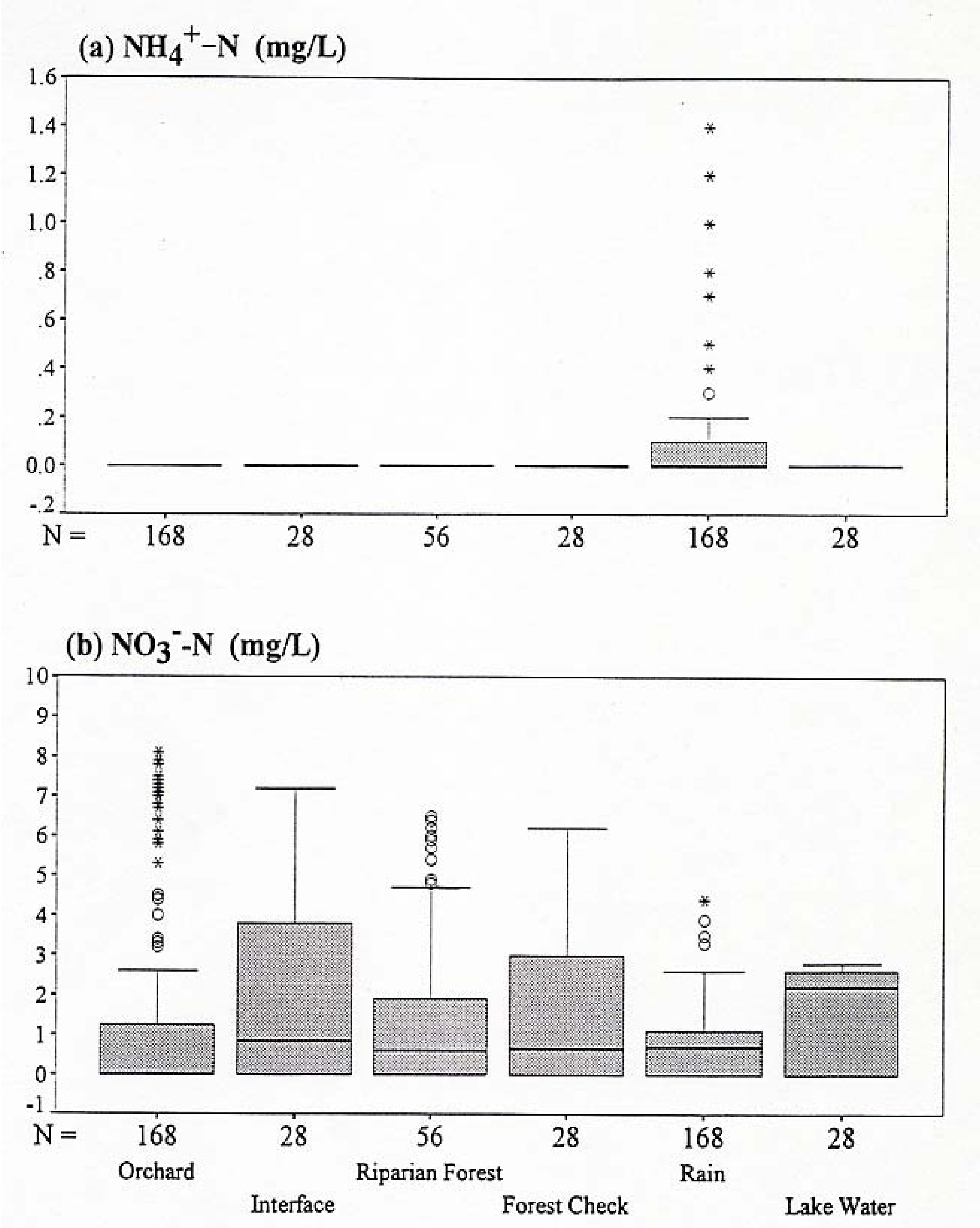


Fig.10. The distribution of ammonium nitrogen and nitrate nitrogen ions in the soil solution collected from different sampling area of Feitsui reservoir near Taipei city.

Notes: The values in the black boxes ranged from 25th % to 75th % of total range. The black bar is medium value (50 % of total range). The values of the bars ranged from minimum value to maximum value. The circle value (O) is the 1.5 times of observation values of black boxes far from the 75th % of the total range values. The star value (*) is the 3 times of observation values of black boxes far from the 75th % of the total range values. N is total sampling number.

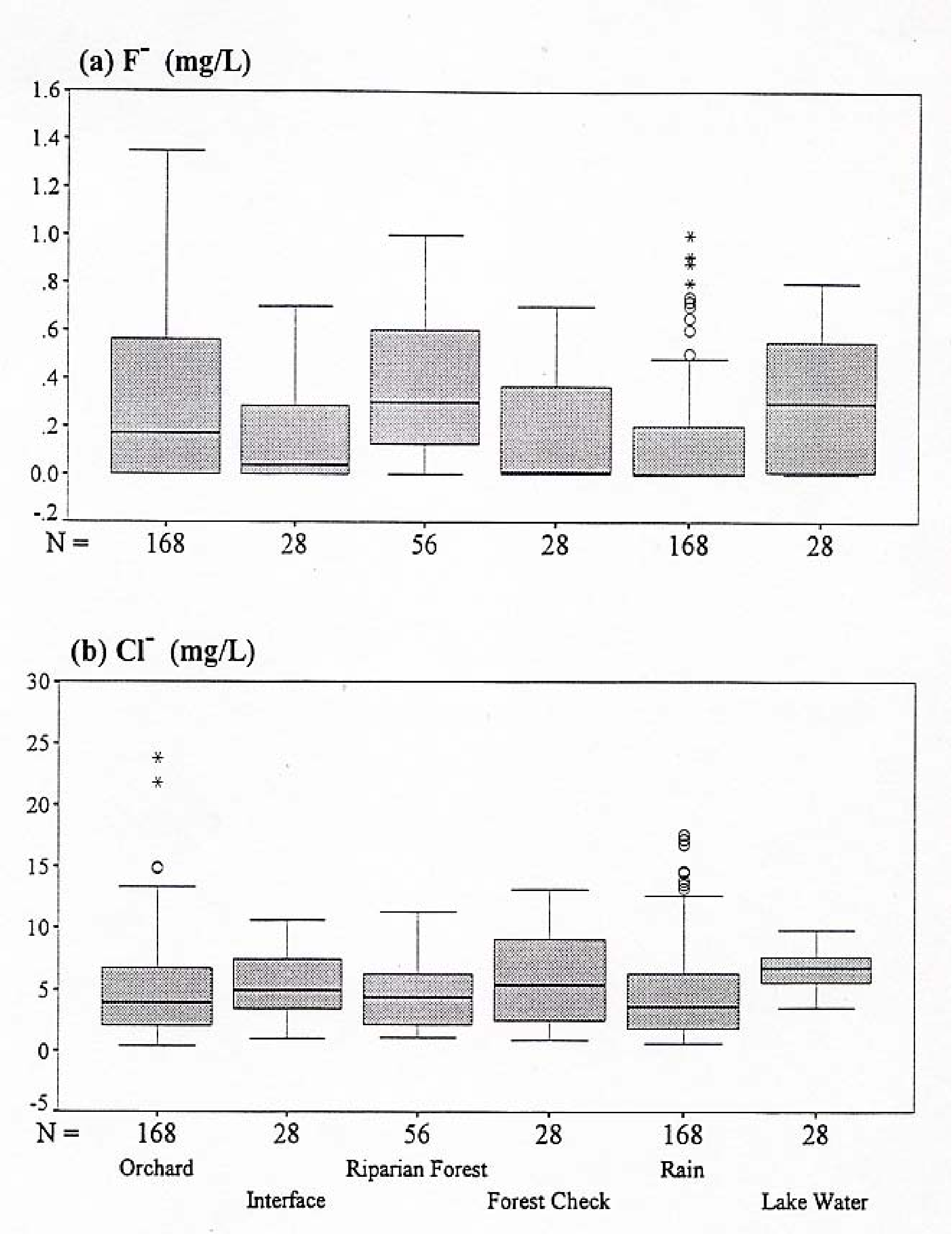


Fig.11. The distribution of fluoride and chloride ions in the soil solution collected from different sampling area of Feitsui reservoir near Taipei city.

Notes: The values in the black boxes ranged from 25th % to 75th % of total range. The black bar is medium value (50 % of total range). The values of the bars ranged from minimum value to maximum value. The circle value (O) is the 1.5 times of observation values of black boxes far from the 75th % of the total range values. The star value (*) is the 3 times of observation values of black boxes far from the 75th % of the total range values. N is total sampling number.

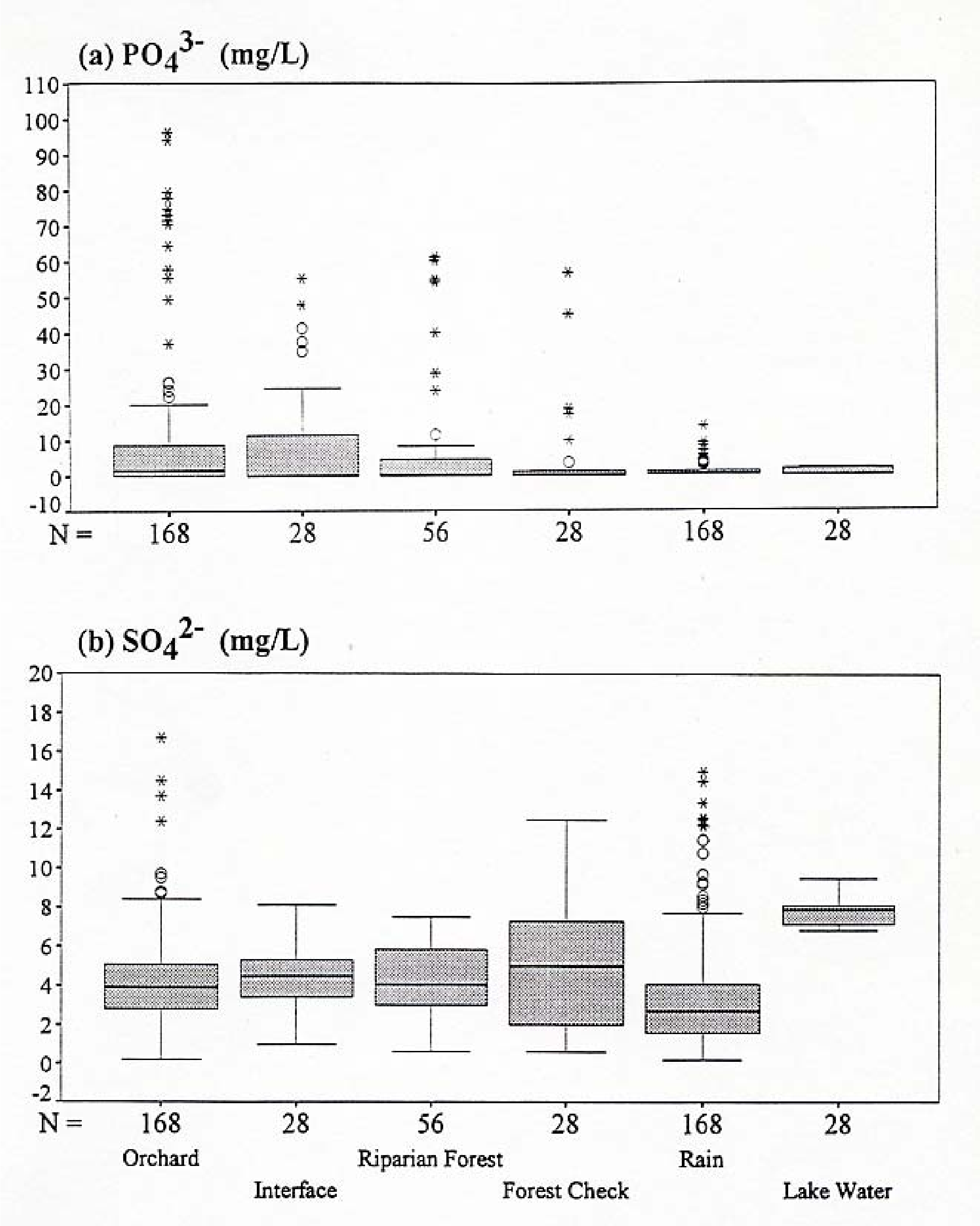


Fig.12. The distribution of phosphate and sulfate ions in the soil solution collected from different sampling area of Feitsui reservoir near Taipei city.

Notes: The values in the black boxes ranged from 25^{th} % to 75^{th} % of total range. The black bar is medium value (50 % of total range). The values of the bars ranged from minimum value to maximum value. The circle value (O) is the 1.5 times of observation values of black boxes far from the 75^{th} % of the total range values. The star value (*) is the 3 times of observation values of black boxes far from the 75^{th} % of the total range values. N is total sampling number.

Using Nitrogen and Oxygen Isotopes to Identify the Sources and Transformations of Nitrate in Groundwater: a Case Study of Groundwater in Nantou Agriculture Region, Taiwan

Tsung-Ren Peng[1,2]

Abstract

This report integrated the concentration, and nitrogen and oxygen isotope compositions of groundwater's NO_3^- in agricultural and urban regions of Ming-Jian, Nantou to illustrate the situation of NO_3^- pollution and the NO_3^- sources and transformation. The results indicate that nearly 54 % of NO_3^- determinations exceeded the criterion of drinking water, especially in agricultural region. The polluted groundwater mainly source NO_3^- from soil nitrogen and chemical fertilizer. The results also indicate the denitrification/assimilation that will reduce the NO_3^- concentration is ambiguous in groundwater of agricultural region, but the assimilation is significant in urban groundwater.

Keywords: NO_3^- pollution, nitrogen and oxygen isotopes, groundwater, Nantou.

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利用氮氧同位素鑑別地下水中硝酸鹽之來源與轉化：以南投農業區地下水為例

彭宗仁 [1,2]

摘　要

本報告整理南投名間農作區與城鎮區地下水的 NO_3^-濃度及其氮、氧同位素組成，以說明此地區地下水 NO_3^-污染狀況及探討地下水中 NO_3^-的來源及轉化。本報告區約有 54 %水樣之 NO_3^-濃度高於飲用水標準，主要發生在農作區內。由氮、氧同位素證據顯示，這些超標地下水中的 NO_3^-主要來自土壤有機物和化學肥料。同位素證據同時顯示農作區地下水中並未發生造成 NO_3^-濃度降低之脫氮或同化作用；反之，城鎮區之地下水則很明顯曾受到同位素分化作用。

關鍵詞：NO_3^-污染、氮氧同位素、地下水、南投

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前言

一般作物生長需要的養分大部分是靠土壤肥力提供，不足部分再以肥料補充；但是，一般而言農民皆有施肥過多的情況。以台灣中部梨山地區爲例，當地農友施用之複合肥料換算成三要素用量，約爲推薦用量之 10 倍 (Lai, 1999)。爲求較佳的收益，農業生產過程需投入肥料。農地對於營養鹽雖有涵容能力，但其承載能力有其限度。施肥超過土壤之負荷量，不僅會對地下水環境造成衝擊，如以受污染之地下水作爲飲用水，則可能危害健康。以硝酸鹽 (NO_3^-) 爲例，飲用含高濃度 NO_3^- (＞50 mg L^{-1}) 的水會導致嬰兒出現「藍嬰症」 (變性血色素病，Cyanosis, blue-tinged blood)，並引發膀胱癌、卵巢癌、胃癌及非霍金氏淋巴瘤等 (Comly, 1945; Feast *et al*., 1998)。因此，各國均訂有相關標準，規定硝酸態氮 (NO_3^--N) 濃度應小於 10 mg L^{-1}，或硝酸鹽 (NO_3^-) 濃度應小於 45~50 mg L^{-1} (Oakes, 1991)。

本報告以南投名間地區爲例，說明該地區地下水硝酸鹽現況；引用資料主要來自彭宗仁等 (2008a；2008b) 之文獻內容。本報告地下水區分爲以農爲主之八卦山台地及位於貓羅溪流域之城鎮區。八卦山台地爲境內主要農作區，土質多爲紅壤；因土壤呈酸性，因此種植作物以茶樹、鳳梨及生薑等耐酸作物爲主。根據名間鄉八卦山台地地下水之水質監測調查分析 (Water Resources Agency, 2001)，地下水中的 NO_3^-濃度在 2001 年調查時已超過飲用水水質標準。

由於不同來源的 NO_3^-有不同的氮和氧同位素特徵值 ($\delta^{15}N$、$\delta^{18}O$)，因此水體 NO_3^-的穩定氮、氧同位素組成已廣泛提供有關地表水和地下水中 NO_3^-可能的來源和轉變的資訊 (Cey *et al.*,1999; Kendall and Caldwell, 1998; Silva *et al.*, 2000; Wassenaar, 1995)。因此，本報告主要藉由農業區與城鎮區地下水 NO_3^-濃度之比較，以了解南投名間地下水受 NO_3^-污染狀況；另一方面，藉由分析地下水中 NO_3^-之氮、氧同位素組成，說明該區地下水中 NO_3^-的可能來源與轉化作用。

材料與方法

地下水樣本

本報告採樣點主要分爲農業區 (agricultural region) 及城鎮區 (urban region) 二區 (圖 1)。農業區地下水井位於八卦台地上，採樣點包括名松段 (Mj1)、新厝 (Mj2)、大坑 (Mj3)、新光 (Mj4)、大庄 (Mj5)、炭寮 (Mj6)、崁腳 (Mj7) 等 7 口公有及私人水井，井深 78 m~200 m。農業區主要種植鳳梨、茶樹、生薑等耐酸作物。城鎮區地下水採樣點包括新街 (Mj8)、萬丹

(Mj9)、新民-1 (Mj10)、新民-2 (Mj11)、雲天宮 (Mj12) 等 5 口公有及私人地下水井，井深約 30 m~120 m。此外，採樣點 Mj13 為雲天宮 (Mj12) 斜坡下方之湧泉。

NO_3^- 濃度分析

NO_3^-之濃度是以紫外光分光光度計法 (NIEA.W419.51A, Environmental Analysis Laboratory 2006) 測定。為排除氫氧根或碳酸根之干擾，在 50 mL 水樣加入 1 mL 的 1N HCl 試液；分別在波長 220 nm 與 275 nm 下，以紫外光分光光度計測定其淨吸光度，再與標準溶液之檢量線比較，求得水樣之 NO_3^--N 濃度。

NO_3^- 氮、氧同位素組成分析

水樣中 NO_3^-之萃取及其氮、氧同位素組成分析主要參照相關文獻 (Chang *et al.*, 1999; Silva *et al.*, 2000; Fan, 2005; Zhan, 2003)。簡言之：水樣通過陰離子交換管柱 (Bio-Rad AG® 1-X8) 以交換萃取水中之 NO_3^-，使用 3N HCl 溶液淋洗管柱並收集濾液，濾液加入 Ag_2O 產生 $AgNO_3$ 溶液，$AgNO_3$ 溶液加入 $BaCl_2$ 及活性碳以去除有機質，以真空冷凍乾燥法製備 $AgNO_3$ 粉末。

$AgNO_3$ 粉末分為二分，一分之 $AgNO_3$ 粉末利用元素分析儀 (element analyzer) 將 $AgNO_3$ 粉末轉換成 N_2，再進入氣相質譜儀 (isotope ratio mass spectrometer, IRMS) 測定其氮同位素組成。另一分之 $AgNO_3$ 粉末置於石英管內並加入約 10 mg 的石墨粉末，並在真空條件下以氫氧焰封燒石英管。此含樣品之石英管在 850℃高溫爐中反應產生 CO_2 氣體，續以氣相質譜儀分析此 CO_2 氣體的氧同位素組成。

質譜儀測定後以 $\delta^{15}N$ 及 $\delta^{18}O$ 代表標本之氮、氧同位素組成，單位以‰表示：

$$\delta = [(R_{sample} - R_{std.}) / R_{std.}] \times 1000 \qquad (1)$$

上式中，δ 代表 $\delta^{15}N$ 或 $\delta^{18}O$，R_{sample} 代表標本之 $^{15}N/^{14}N$ 或 $^{18}O/^{16}O$ 比值，$R_{std.}$代表標準品之 $^{15}N/^{14}N$ 或 $^{18}O/^{16}O$ 比值。本報告使用台大地質科學系之質譜儀進行氮同位素組成分析，質譜儀型號為 Finnigan DELTA plus；氧同位素組成分析則使用中央研究院地球科學研究所之質譜儀，型號為 VG SIRA10。

結果

NO_3^-濃度

由彭宗仁等 (2008a) 分析顯示，農業區地下水 NO_3^-的濃度平均值範圍為 43.9~225 mg L^{-1}

(圖 2a)，平均值約 111 mg L^{-1}；其中以大坑 (Mj3) 之濃度最高，平均值爲 225 ± 9.3 mg L^{-1} (範圍在 202~237 mg L^{-1})；最低爲崁腳 (Mj7)，平均值爲 43.9 ± 2.1 mg L^{-1} (濃度範圍爲 40.8~47.0 mg L^{-1})。農業區地下水 NO_3^-濃度幾已超過飲用水水質標準，雖崁腳 (Mj7) 地下水 NO_3^-濃度平均值雖未超過飲用水標準，但 2005 年 5 月之測值 (47 mg L^{-1}) 則超過標準。由於這些採樣井周圍種植大面積的鳳梨、茶樹及生薑；因此，地下水含過高之 NO_3^-應與農業活動有密切關聯。根據農業區地下水 NO_3^-濃度的月歷線圖顯示 (圖 2a)，大多數地下水之 NO_3^-濃度並未有明顯之季節性變化，僅大庄 (Mj5) 於 2005 年 2 月、5 月及 6 月之 NO_3^-濃度分別爲 126, 125, 139 mg L^{-1}，較平均值 162 ± 23.6 mg L^{-1} 爲低。

城鎮區地下水 NO_3^-的濃度平均值範圍爲 6.02~63.9 mg L^{-1} (圖 2b)，平均值約 27.2 mg L^{-1}；其中以萬丹 (Mj9) 採樣井濃度最高，平均值達 63.9 ± 4.7 mg L^{-1} (範圍在 54.8–68.7 mg L^{-1})，已超過飲用水水質標準。整體而言，在 2004 年 6 月至 2005 年 9 月之分析期間，城鎮區地下水約有 54%的地下水之 NO_3^-濃度測定超過飲用水水質標準。

NO_3^-之氮、氧同位素組成

由彭宗仁等 (2008b) 分析顯示，農作區地下水中 NO_3^-之 $\delta^{15}N$ 值範圍爲 3.7~8.5 ‰ (圖 3)，以炭寮 (Mj6) 最重，平均值爲 6.8±1.2 ‰；新厝 (Mj2) 最輕，平均值爲 4.7±1.0 ‰。城鎮區地下水 NO_3^-之 $\delta^{15}N$ 值範圍爲 3.1~11.0 ‰，以湧泉 (Mj13) 爲最重，平均值爲 9.8±0.4 ‰；雲天宮 (Mj12) 次之，平均值爲 9.1±2.0 ‰，新街 (Mj08) 爲最輕，平均值爲 4.3±1.6 ‰。新民-1 (Mj10) 與鄰近的新民-2 (Mj11) 兩者之差異並不大，平均值分別爲 4.5±0.8 ‰及 4.9±1.8 ‰。總體而言，農作區地下水其 $\delta^{15}N$ 值呈現出旱季 (12 月及 3 月) 較重而雨季 (6 月及 9 月) 較輕的季節性差異現象，但城鎮區地下水 $\delta^{15}N$ 值的季節性差異現象則不明顯。

本報告所有地下水各季 NO_3^-之 $\delta^{18}O$ 值與 $\delta^{15}N$ 值之分布如圖 4 所示。農作區地下水 NO_3^-之 $\delta^{18}O$ 值範圍爲-7.6 至 3.5 ‰。以平均值而言，炭寮 (Mj6) 之值最重，其平均值爲-1.7±4.9 ‰；大庄 (Mj5) 最輕，平均值爲-6.2±0.1 ‰。整體而言，農作區地下水 NO_3^-之 $\delta^{18}O$ 值，除大坑 (Mj3) 之雨、旱季節性差異較爲明顯外，其餘地下水並不明顯。城鎮區地下水 NO_3^-之 $\delta^{18}O$ 值範圍爲-7.3 至 7.6‰。以平均值而言，最輕者爲新民-2 (Mj11)，平均值爲-4.9±2.4 ‰；最重者爲雲天宮 (Mj12)，平均值爲 0.1±5.7 ‰。整體而言，除新街 (Mj8) 及新民-1 (Mj10) 外，其餘各點地下水 NO_3^-之 $\delta^{18}O$ 值皆呈現出旱季較雨季爲重的現象。

以整區而言，農作區地下水 NO_3^-之 $\delta^{18}O$ 值季節性差異並不明顯；而城鎮區則大都顯示出旱季較重、雨季較輕的現象。

討論

NO_3^-污染的程度與分佈

由彭宗仁等 (2008a) 整理名間地區地下水 NO_3^-濃度平均值之等值線分布圖顯示 (圖 5)，高 NO_3^-濃度之地下水分布在八卦山台地之農業區。另一方面，NO_3^-濃度似有隨高程之降低 (地下水之流向)，而向城鎮區遞減之趨勢。然而，在 NO_3^-等值線 50 mg L^{-1} 與 100 mg L^{-1} 間之區域較無規律分佈 (圖 5)。

由圖 5 顯示，農業區地下水在向城鎮區流動過程中，可能同時受到面狀式與點狀式之污染源的影響。由於八卦山農業區地層以粗質地的礫石為主，缺乏較厚的阻水層，地下水上下流通未受限制。因此，地表的含氮物質隨著水流從地表縱向下滲至含水層後造成 NO_3^-濃度上升，為面狀式污染源 (圖 5)。之後在往下游移動時，隨著水流移動距離增加，因為水力分散作用造成 NO_3^-濃度逐漸降低；然而在部分下游地點發生點源污染，因此造成在下游處仍有點狀高 NO_3^-濃度的現象 (圖 5)。

城鎮區為人口較為密集的區域，當地產生的家庭污水相對較多，因此，家庭污水可能未經適當處理即排入河川，進而污染地下水。例如，以本報告城鎮區而言，新民-1 (Mj10) 採樣井為私人抽水井供養殖鱒魚用，養殖業主要污染源主要來自養殖廢水及養殖飼料的排放，部分養殖業大量排放污水 (含池魚的排泄物以及大量未被吸收而溶入池塭的飼料)，易使污染物透過土壤及附近的河川而滲入地下水。

各採樣井地下水 NO_3^-濃度之季節性變異程度可由濃度平均值之變異係數 (coefficient of variation, $CV = \frac{SD}{\bar{x}} \times 100\%$) 來指示。除雲天宮 (Mj12)、湧泉 (Mj13) 外，大部分採樣井地下水 NO_3^-濃度平均值之 CV 值皆在 20%以下，顯示 NO_3^-濃度之季節性變化不明顯，意指地下水之 NO_3^-有穩定的來源。同時，在研究期間，約有 54 %城鎮區地下水之 NO_3^-濃度測定超過飲用水水質標準，而在農業區之地下水幾乎皆已超過飲用水水質標準 (圖 2)。因此，南投名間地區之地下水受 NO_3^-污染之情況甚為明顯 (彭宗仁等，2008a)。

NO_3^-之可能來源

根據台灣一般常用肥料之 $\delta^{15}N$ 特徵值 (彭宗仁等，2004)，各類化學肥料之 $\delta^{15}N$ 值分佈範圍相近，如硫酸銨之 $\delta^{15}N$ 平均值為-0.8±0.7 ‰，尿素之之 $\delta^{15}N$ 平均值為-0.8±0.3 ‰；而動物性糞肥如乾雞糞之 $\delta^{15}N$ 平均值為 17.5±0.3 ‰，此肥料之 $\delta^{15}N$ 特徵值與國外文獻相符 (Kendall and Caldwell, 1998)。名間鄉農作區之旱田作物主要施肥以禽畜糞堆肥及化學肥料為

主。本報告所有地下水之 $\delta^{15}N$ 值與相關氮源之特徵值範圍比較 (圖 3) (彭宗仁等，2008b)，農作區地下水 (Mj1~Mj7) 其 $\delta^{15}N$ 值範圍爲 3.7 至 8.5 ‰，主要分布在化學肥料與土壤氮源特徵值間，但部分樣品似有受到動物排泄物源的影響而顯示出較重的 $\delta^{15}N$ 值。

城鎮區地下水 (Mj8~Mj13)，其 $\delta^{15}N$ 值範圍爲 3.1 至 11.0 ‰ (圖 3)；在人口密度較高的新街、萬丹以及靠近濁水溪河岸的新民-1、新民-2 等四井地下水，其 $\delta^{15}N$ 範圍爲 3.1 至 7.4 ‰，其分布範圍與農作區地下水之情況類似。而雲天宮 (Mj12) 及湧泉 (Mj13) 兩者有較重之 $\delta^{15}N$ 值，分布範圍分別爲 6.3 至 11 ‰與 9.4 至 10.2 ‰。以圖 3 之比對而言，雲天宮地下水 NO_3^- 之來源與土壤氮源和動物排泄物源有關，但似乎受到動物排泄物源的影響較大；湧泉則顯示主要受到動物排泄源影響。

由地下水 NO_3^- 之氮、氧同位素組成分布與文獻特徵值比較顯示 (圖 4) (彭宗仁等，2008b)，大部份地下水之 NO_3^- 爲土壤有機氮、化學肥料或動物排泄物經硝化作用產生而來；城鎮區雲天宮 (Mj12) 和湧泉 (Mj13) 水樣之 NO_3^- 與動物排泄物經硝化作用產生有關。

地下水中 NO_3^- 之轉化作用

地下水 NO_3^- 之濃度及氮氧同位素組成可能受到轉化作用影響。通常 NO_3^- 之轉化會造成同位素發生分化作用，此同位素分化作用過程是以雷利方程式 (Rayleigh equation) 之指數曲線變化表示，而 NO_3^- 之濃度與 $\delta^{15}N$ 值之關係以下述關係式表之 (Kendall and Caldwell, 1998; Mariotti *et al.*, 1988)：

$$\delta^{15}N = \delta^{15}N_0 + \varepsilon \ln NO_3^- \qquad (2)$$

上式中 $\delta^{15}N_0$ 爲初始之氮同位素組成；NO_3^- 與 $\delta^{15}N$ 分別代表在 t 時間之 NO_3^- 濃度及其氮同位素組成；ε 爲富集係數 (enrichment factor)，即 NO_3^- 之轉化作用造成氮同位素發生同位素分化作用，導致反應前、後 NO_3^- 的 $\delta^{15}N$ 值改變。

在水文系統中，NO_3^- 之輸入主要爲含氮物質經硝化作用或混合作用而進入，NO_3^- 之輸出則是因藻類吸收 NO_3^- 的同化或 NO_3^- 的脫氮作用造成。所以，相關 NO_3^- 轉化作用所造成同位素分化的現象與程度可藉由 $\ln NO_3^-$ 與 $\delta^{15}N$ 間之關係來了解 (Kendall and Caldwell, 1998; Mariotti *et al.*, 1988)。發生同化作用時，因藻類會優先攝取 ^{14}N 進入有機體，造成水體中 NO_3^- 濃度會降低；同時，殘餘的 NO_3^- 會因富集 ^{15}N 使得 $\delta^{15}N$ 值變重 (Shear and DeBruyn, 1986)。脫氮作用爲 NO_3^- 被還原成 N_2 的過程，由於微生物會優先利用較輕 (^{14}N) 之 NO_3^- 而產生具較輕 $\delta^{15}N$ 特徵之 N_2 等氣體，但造成 NO_3^- 濃度降低，同時也使殘餘之 NO_3^- 具較重之 $\delta^{15}N$ 值 (Knowles, 1982)。因此，同化與脫氮作用皆會降低地下水 NO_3^- 濃度，並使殘餘之 NO_3^- 具較重

之 $\delta^{15}N$ 值。若發生同化或脫氮作用，則 (2) 式中之 ε 值則為小於零。

由彭宗仁等 (2008b) 分析顯示，農作區地下水 $\delta^{15}N$ 與 $\ln NO_3^-$之關係式為 (圖 6a)：

$$\delta^{15}N = 1.3 \ln NO_3^- - 0.4，R^2 = 0.29 \quad (3)$$

(3)式中 ε 值為正值，且為中度相關 (r＝0.54)。因此，農作區地下水並沒有受同化或脫氮作用的影響。

相對而言，城鎮區地下水 $\delta^{15}N$ 與 $\ln NO_3^-$之關係式為 (圖 6b)：

$$\delta^{15}N = -1.9 \ln NO_3^- + 11.9，R^2 = 0.46 \quad (4)$$

上式中 ε 值為負值，且為中度負相關 (r = -0.68)，顯示該區地下水受到同化作用或脫氮作用的影響，造成水體 NO_3^-濃度降低而 $\delta^{15}N$ 值變重。由於同化與脫氮作用皆會降低水體中 NO_3^-濃度、並造成殘餘 NO_3^-的 $\delta^{15}N$ 值變重。為鑑別此二種輸出作用，則必須先了解同化與脫氮作用發生的條件。

一般水體發生脫氮作用的理想條件為：pH 值在 6.2 至 10.2；Eh 在-200~665 mv；溫度範圍在 0~50℃；DO 濃度須小於 2 mg L^{-1} (Thomasson *et al.*, 1991)。假設發生脫氮作用之 DO 濃度之上限值為 2 mg L^{-1} (Gillham and Cherry, 1978; Karr *et al.*, 2001; Thomasson *et al.*, 1991)，本報告農作區地下水之 DO 濃度範圍在 3.5~7.3 mg L^{-1} (圖 7)，是屬於較為曝氣的水體 (Mengis *et al.* 1999)。相較而言，城鎮區地下水之 DO 濃度，除湧泉 (Mj13) 較低外 (平均值為 1.4±0.5 mg L^{-1}，圖 7)，其餘地下水之 DO 濃度皆大於 2 mg L^{-1}，範圍在 2.9~7.0 mg L^{-1}。圖 7 顯示除湧泉外，其餘地下水發生顯著脫氮作用的機會較小。此外，Kendall and Caldwell (1998) 指出，如果地下水發生脫氮作用，其 ε 值的分佈範圍在-40 至-5 ‰；Fukada *et al.* (2003, 2004) 指出脫氮作用時 ε 值約為-13.6 ‰；而 Selier (2005) 指出發生脫氮作用時 ε 值為-5.7 ‰。因此，由式 (4) 中之 ε 值顯示為-1.9 ‰，說明城鎮區地下水並沒有發生顯著之脫氮作用。

當發生脫氮作用時，微生物將 NO_3^-中的 N 轉變成 N_2，並生成具較輕 $\delta^{15}N$ 值的氮氣，則殘餘的 NO_3^-會較富集 ^{15}N (較重之 $\delta^{15}N$ 值)；在此同時，也將 NO_3^-中的 O 原子，轉變成其他的含氧化合物，亦使殘餘的 NO_3^-亦會較富集 ^{18}O (Aravena and Robertson, 1998; Böttcher *et al.*, 1990; Mariotti *et al.*, 1988; Smith *et al.*, 1991)。因此，脫氮作用會同步富集的氮、氧同位素會組成，並形成一約 $\delta^{15}N/\delta^{18}O \fallingdotseq 1.3 \sim 2.1$ 的線性關係 (Devito *et al.*, 2000; Fukada *et al.*, 2003, 2004; Mariotti *et al.*, 1982; Mengis *et al.*, 1999)；因此，亦可由此線性關係來評估地下水中是否發生脫氮作用 (Cey *et al.*, 1999; Mosier and Schimel, 1993; Wassenaar, 1995)。而由圖 8 顯示 (彭宗仁等，2008b)，城鎮區地下水 NO_3^-之氮、氧同位素組成約呈 1：1 的線性關係 ($\delta^{18}O = 0.92 \times$

$\delta^{15}N - 9.2$)；所以，城鎮區地下水似乎未發生明顯之脫氮作用。

由水生植物 (藻類) 對 NO_3^-吸收的同化作用所造成之同位素分化結果顯示其 ε 值範圍在 -27 至 0 ‰ (Kendall and Caldwell, 1998; Kohl and Shear, 1980; Shear and DeBruyn, 1986)，而式 (4) 中的 ε 值為-1.9。因此，由圖 7 顯示在城鎮區地下水，NO_3^-濃度降低伴隨著 $\delta^{15}N$ 值變重的現象。應是與水生植物對 NO_3^-吸收之同化作用有關 (彭宗仁等，2008b)。

結論

由本報告地下水 NO_3^-之濃度整理顯示，名間地區農作區地下水皆已超過飲用水水質標準且季節性變化不大，應是受到長期農業活動的影響。由氮、氧同位素組成數據顯示，名間地區地下水 NO_3^-主要來自不同氮源經硝化作用所產生之硝酸鹽。其中，農作區受污染的地下水之 NO_3^-主要源自土壤有機氮、化學肥料，但沒有受到相關 NO_3^-轉化作用所影響。城鎮區地下水 NO_3^-的來源主要為土壤有機氮、動物排泄物，但其 NO_3^-濃度多未超過標準值，此現象應與該區發生顯著之同化作用而降低其 NO_3^-濃度有關。

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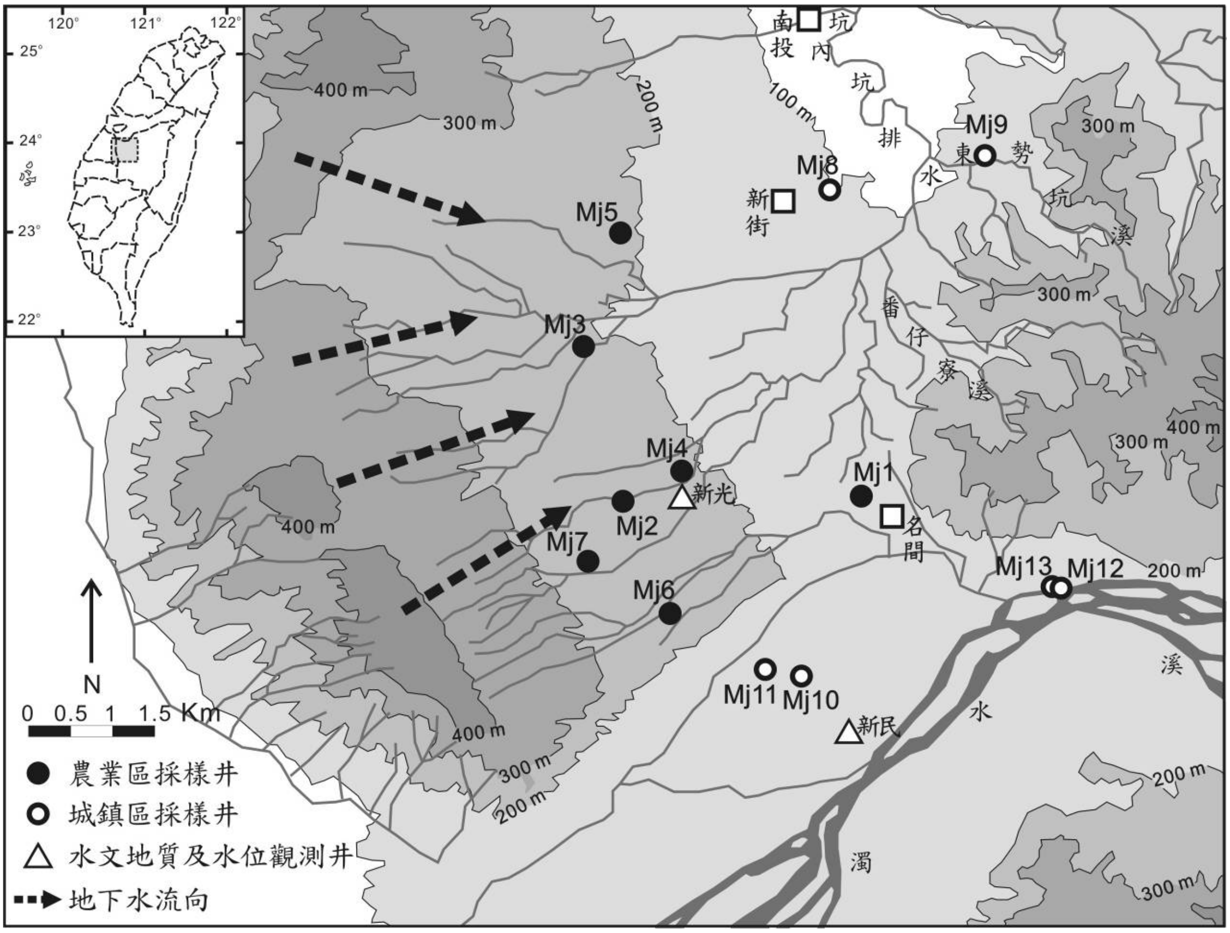

圖 1. 名間地區地形圖及本報告採樣井位置圖。摘自彭宗仁等(2008a)。

Fig. 1. Map showing the topography of Ming-Jian area and the localities of sampling wells of this study.

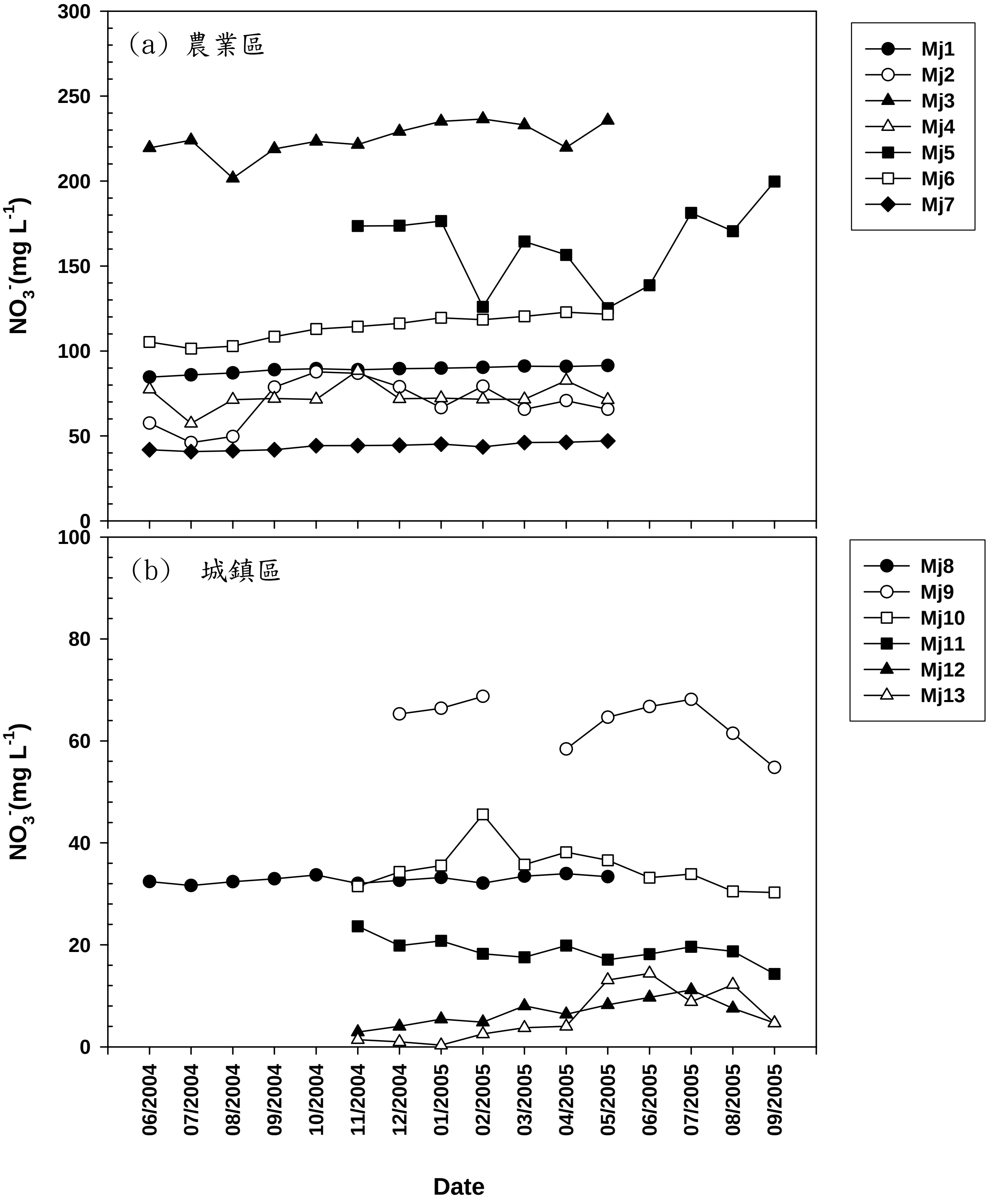

圖 2. 各採樣井地下水之 NO_3^-濃度各月變化圖。摘自彭宗仁等(2008a)。

Fig. 2. Monthly variation of NO_3^- concentration in respective groundwaters.

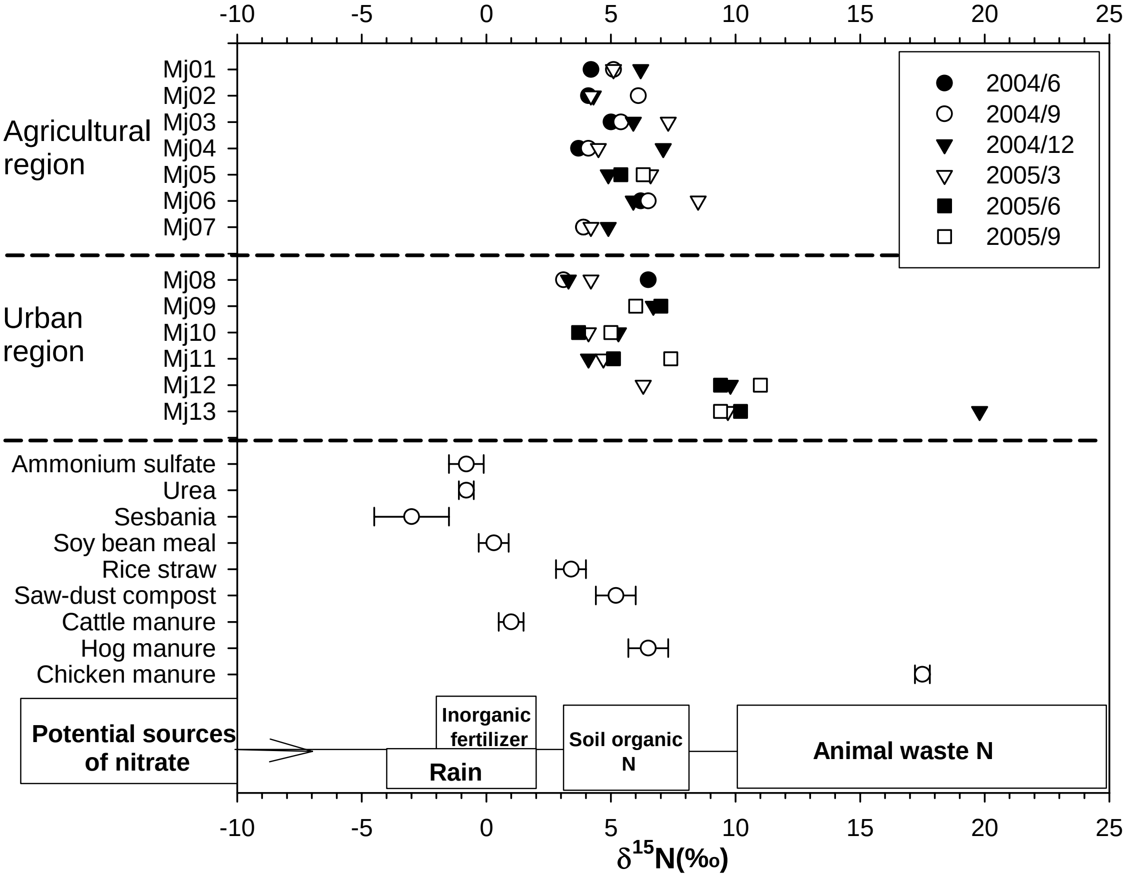

圖 3. 本報告各水體中 NO_3^-之 $\delta^{15}N$ 特徵值分布。摘自彭宗仁等(2008b)。

Fig. 3. Distribution of $\delta^{15}N$ of water NO_3^- in this study.

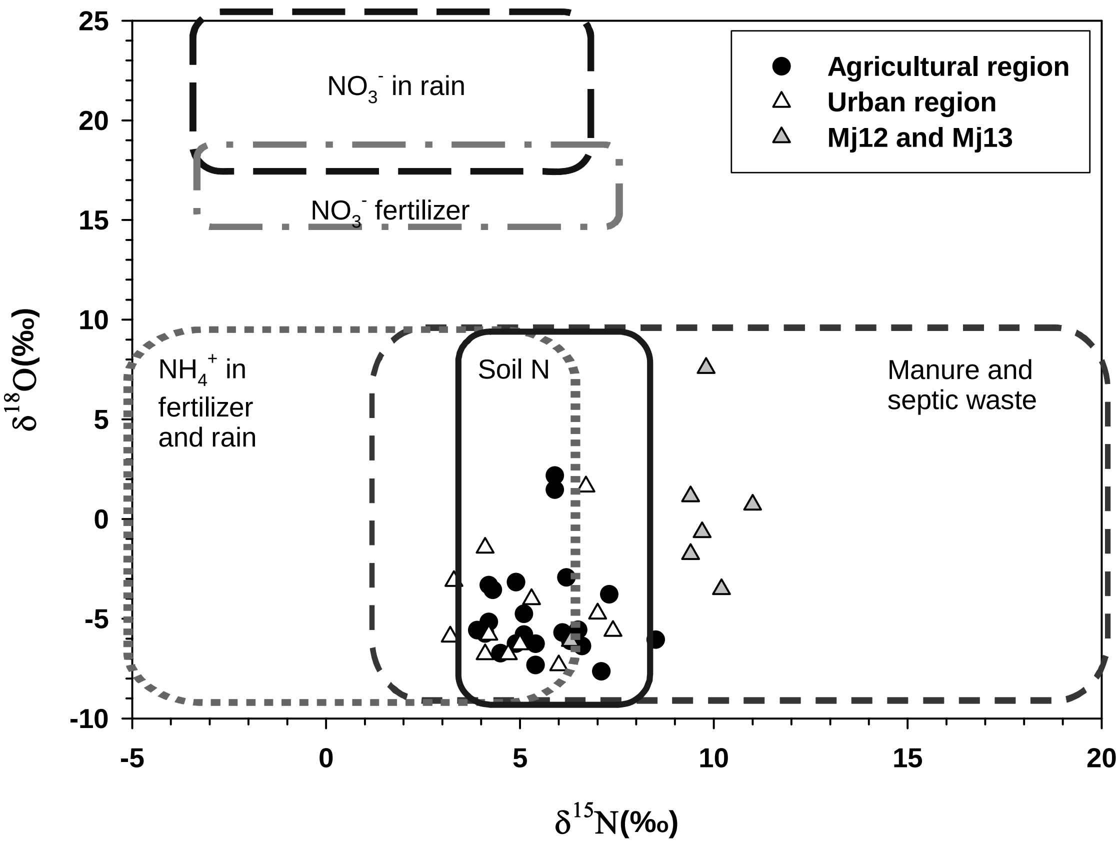

圖 4. 本報告各水體 NO_3^-之氮、氧同位素組成分布圖。摘自彭宗仁等(2008b)。

Fig. 4. Plot of $\delta^{15}N$ vs. $\delta^{18}O$ of water NO_3^- in this study.

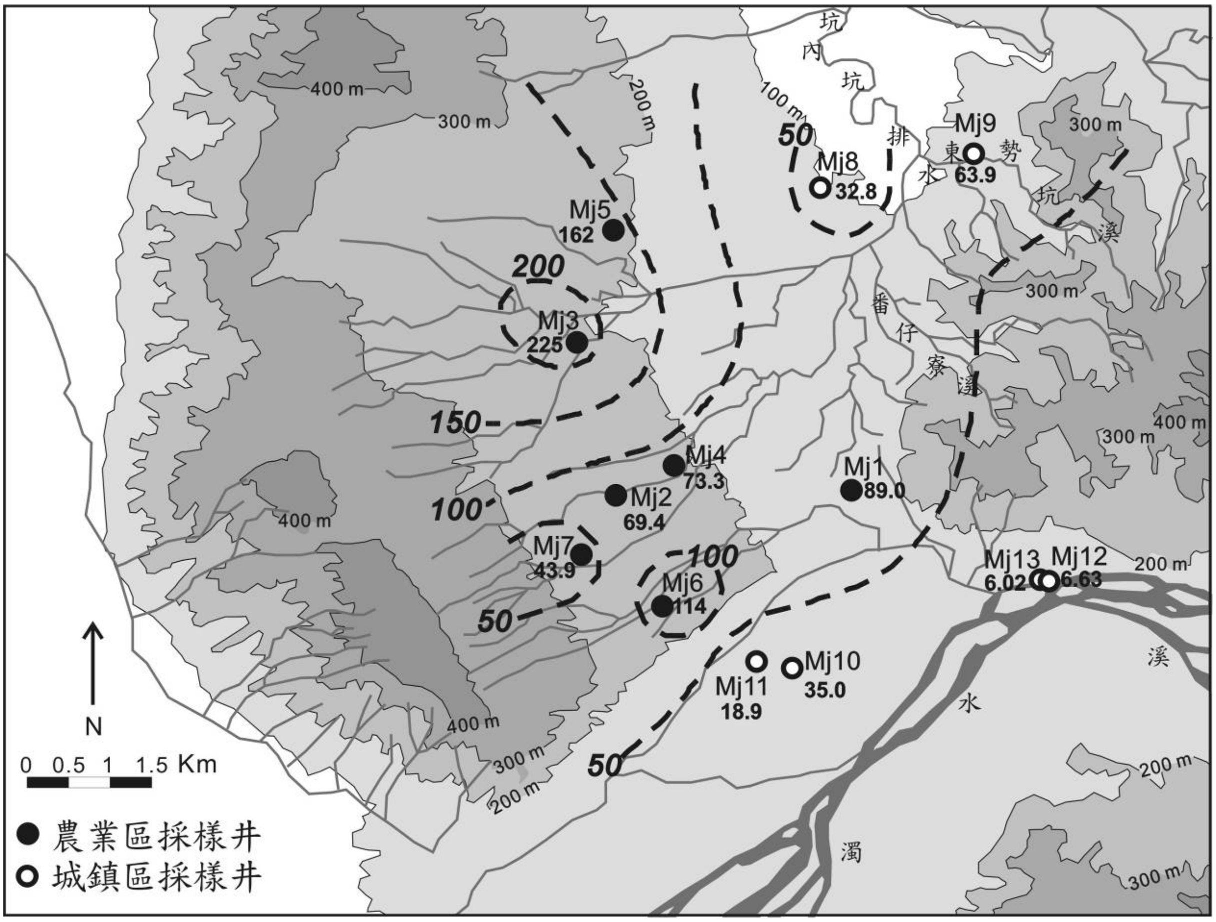

圖 5. 名間地區地下水 NO_3^-濃度 (mg L^{-1}-NO_3^-) 分布變化圖。摘自彭宗仁等(2008a)。

Fig. 5. The NO_3^- contour map of groundwater in Ming-Jian area. (Unit in mg L^{-1}-NO_3^-).

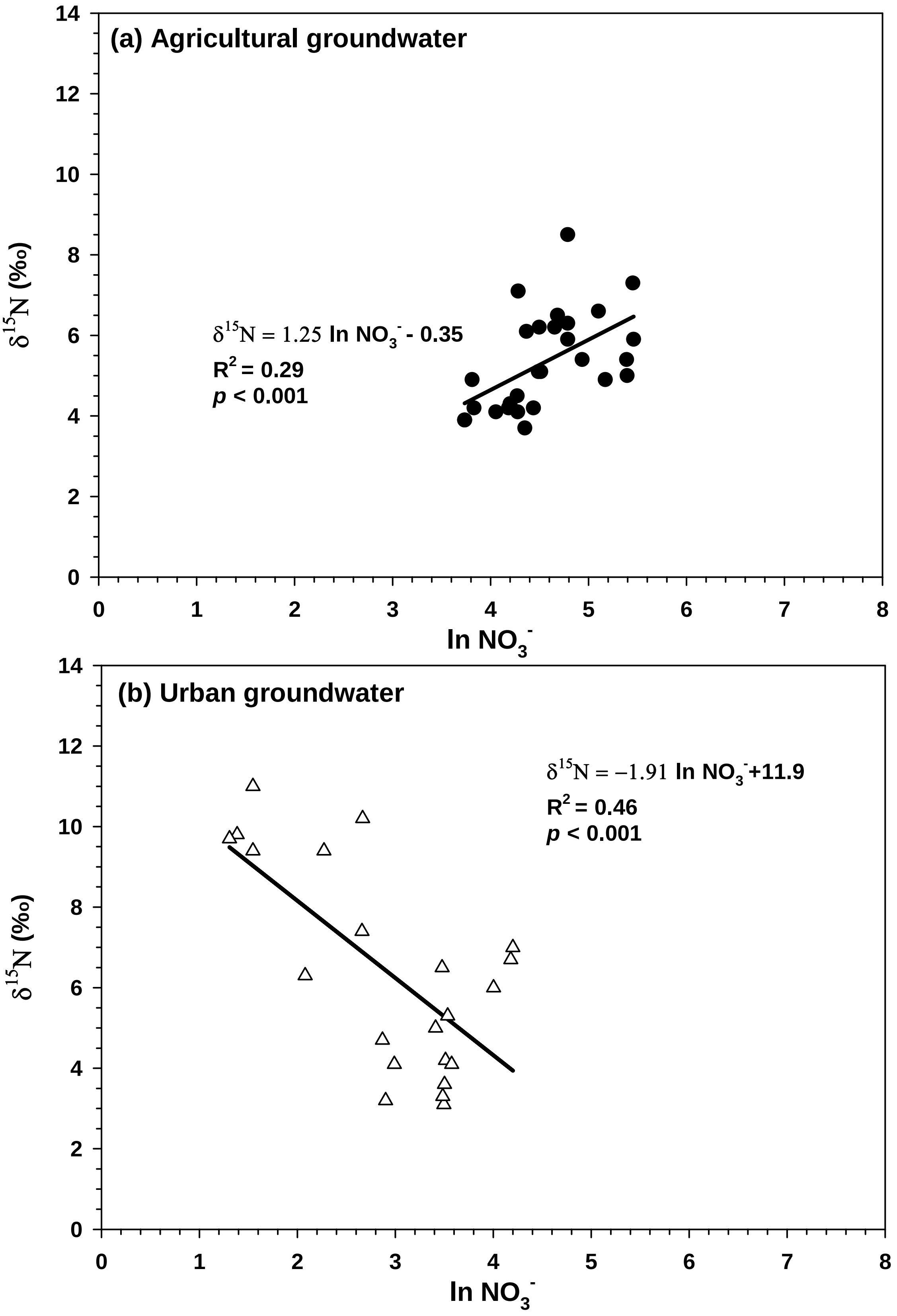

圖 6. 本報告地下水中 NO_3^-之 $\delta^{15}N$ 與 $lnNO_3^-$之關係圖。摘自彭宗仁等(2008b)。

Fig. 6. The relation between $lnNO_3^-$ and $\delta^{15}N$ of water NO_3^- in this study.

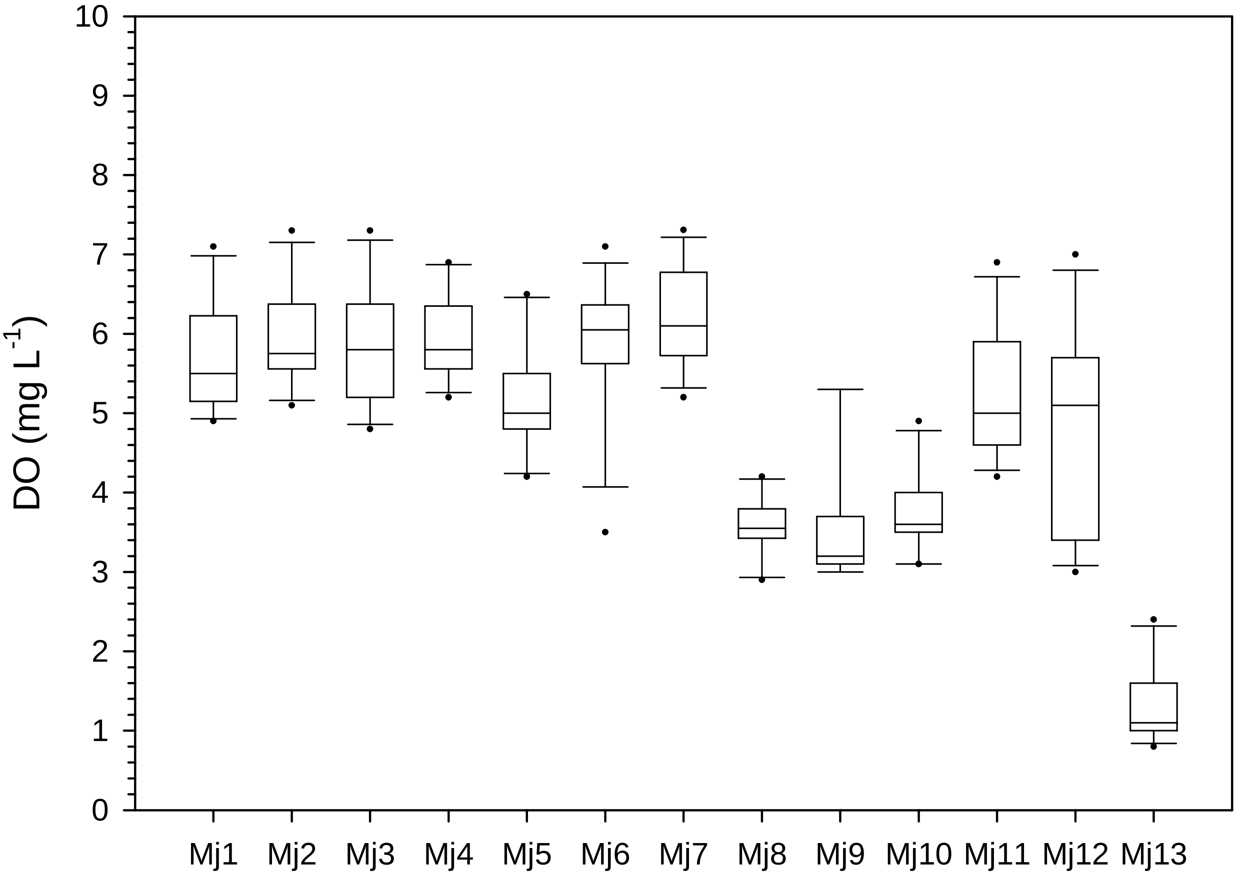

圖 7. 各採樣井地下水之 DO 濃度盒形圖

Fig. 7. Boxplot of the DO concentration of groundwaters during the studied period.

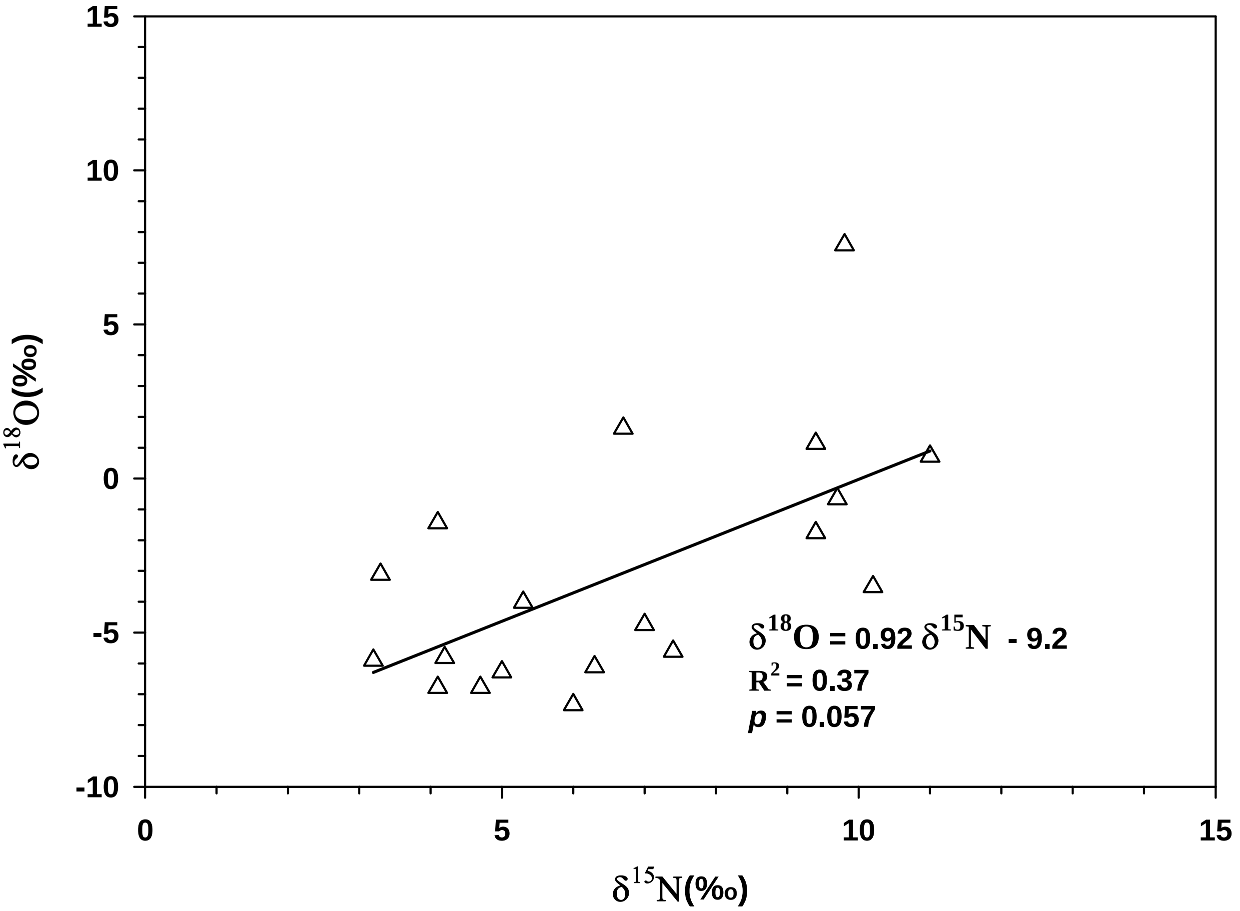

圖 8. 城鎮區地下水 NO_3^-之氮、氧同位素組成分布圖。摘自彭宗仁等(2008b)。

Fig. 8. Plot showing the relation between lnNO_3^- and $\delta^{15}N$ of groundwater NO_3^- in urban region.

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